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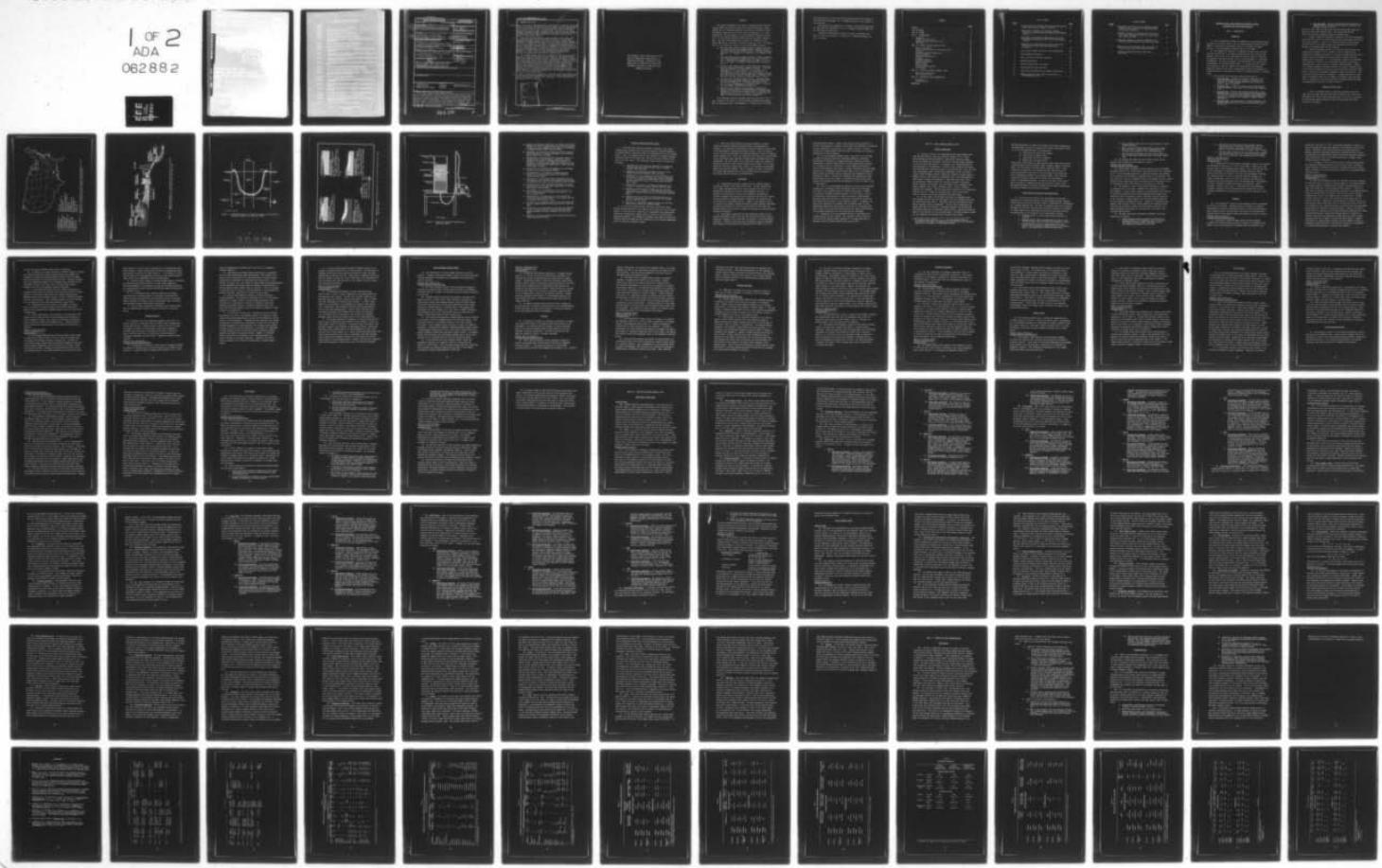
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CONFINED DISPOSAL AREA EFFLUENT AND LEACHATE CONTROL (LABORATOR--ETC(U))
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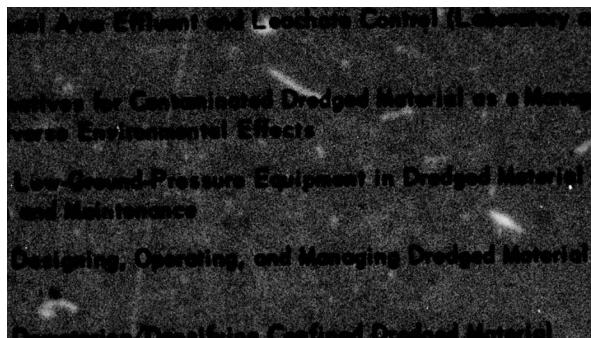
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20. ABSTRACT (Continue on reverse side if necessary and identify by block number) This report summarizes the findings of five work units concerned with the impact of dredged material disposal in confined land disposal areas. Three of the work units dealt with active disposal operations at 11 sites; impact was assessed by comparing the quality of influents and effluents at each site with background surface receiving water. Two work units are discussed which evaluated the impact of confined disposal area leachates on groundwaters. The leachate studies included laboratory column elutions (3- to 9-month period) of	(Continued)		<i>cont'</i>

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20. ABSTRACT (Continued).

CONT. → each of five types of dredged material overlying one of two different soils; four of the 11 field sites were also monitored for changes in leachate and groundwater quality (four onsite, four offsite monitoring, and two offsite background sampling locations at each site) in a 9-month study. The field sites included freshwater (riverine and lake) and brackish water (estuarine and ship channel) dredging environments located in geographical areas where contamination problems were anticipated. The dredged material and environmental features varied greatly at the different sites.

Data from the active disposal studies showed that in most cases soluble concentrations of most chemical constituents were very low. Only soluble manganese and ammonia nitrogen levels failed to meet most criteria. The concentrations of chlorinated hydrocarbons, most trace metals, and total phosphorus in unfiltered effluent water also failed to meet most water quality criteria. Thus, it appears that efficient removal of suspended solids before effluent discharge is necessary to meet these regulatory guidelines. The presence of thick, actively growing vegetation in disposal areas seemed to improve the removal of suspended solids and soluble nutrients (ammonium nitrogen and soluble phosphorus). Salinity levels in effluents from saline dredged material disposal sites should impact low-salinity receiving waters.

The leachate studies suggest that the disposal of brackish water dredged material in upland disposal areas may render subsurface water unsuitable for public water supply or irrigation purposes. Leachates from both the freshwater and brackish water disposal sites usually contained levels of ammonia nitrogen, iron, and manganese that exceeded drinking water standards. The freshwater site also contained an average nitrate nitrogen level that exceeded guidelines. Leachate data from background and offsite monitoring wells showed that high levels of some chemical constituents, especially iron and manganese, were governed by local environmental conditions (i.e. Eh and pH) present in the surrounding soils. This strongly suggests that the physical and chemical properties of the soils underlying a prospective site be considered in context with the nature of the dredged material.

→ It is suggested that guidelines for evaluation of potential disposal sites be developed in a stepwise progression that will not require complete execution of the total program to determine site suitability. A short discussion of an evaluatory scheme is included.

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PREFACE

This report synthesizes the results of Dredged Material Research Program (DMRP) Environmental Impacts and Criteria Development Project (EICDP) Task 2D, "Confined Disposal Area Effluent and Leachate Control." The DMRP was sponsored by the Office, Chief of Engineers, U. S. Army (DAEN-CWO-M), and was assigned to the Environmental Laboratory (EL) of the U. S. Army Engineer Waterways Experiment Station (WES), Vicksburg, Miss. The report was prepared under Contract No. DACW39-76-C-0171 between WES and the University of Southern California (USC), Los Angeles, Calif., and is based on results of the following DMRP work units:

- a. Work Unit 2D01, Technical Report D-78-24, "Physical and Chemical Characterization of Dredged Material Influentes and Effluents in Confined Land Disposal Areas." Principal investigators were R. E. Hoeppel, T. E. Myers, and R. M. Engler of EL, WES.
- b. Work Unit 2D02, Technical Report D-78-20, "A Study of Leachate from Dredged Material in Upland Areas and/or in Productive Uses." Principal investigators were J. L. Mang, J. C. S. Lu, R. J. Lofy, and R. P. Stearns of SCS Engineers, Long Beach, Calif.
- c. Work Unit 2D03, "Monitoring of Trace Constituents During PCB Recovery Dredging Operations, Duwamish Waterway." Principal investigators were J. N. Blazevich, A. R. Gahler, G. J. Vasconcelos, R. H. Rieck, and S. V. W. Pope of the U. S. Environmental Protection Agency, Region X Laboratory, Surveillance and Analysis Division, Seattle, Wash. (Results of this work unit are also presented in Technical Report D-78-24.)
- d. Work Unit 2D04, Technical Report D-78-16, "Characterization of Confined Disposal Area Influent and Effluent Particulate and Petroleum Fractions." Principal investigators were J. C. S. Lu, B. Eichenberger, M. Knezevic, and K. Y. Chen of the Environmental Engineering Program, USC.
- e. Work Unit 2D05, Technical Report D-78-43, "Physical and Chemical Characterization of Dredged Material Sediments and Leachates in Confined Land Disposal Areas." Principal investigators were K. Y. Yu, K. Y. Chen, R. D. Morrison, and J. L. Mang of the Environmental Engineering Program, USC.

This synthesis report was prepared under the direction of Dr. K. Y. Chen, Professor and Director of the Environmental Engineering Program, USC, in cooperation with B. Eichenberger, J. L. Mang, and R. E. Hoeppel.

The participation of M. Knezevic and N. Brovko during various stages of the study are acknowledged. Ms. C. McMahon assisted in the preparation of this report.

The contract was managed by R. E. Hoeppel, Research Microbiologist, EL, WES, under the supervision of Dr. R. M. Engler, Manager, EICDP, and Dr. John Harrison, Chief, EL.

Directors of WES during the period of report preparation were COL G. H. Hilt, CE, and COL J. L. Cannon, CE. Technical Director was Mr. F. R. Brown.

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CONFINED DISPOSAL AREA EFFLUENT AND LEACHATE CONTROL
(LABORATORY AND FIELD INVESTIGATIONS)

PART I: INTRODUCTION

Background

1. The investigations reviewed and summarized in this synthesis report were part of an overall effort under the Dredged Material Research Program (DMRP) to evaluate the environmental impacts of disposal of dredged material. The investigations comprised Task 2D of the DMRP.

2. The studies were important in assessing the short- and long-term potential for mobility of different contaminants placed in confined land disposal areas as well as in evaluating the possible impact of effluents and leachates from these disposal areas on receiving waters or underlying groundwaters. The data were needed to provide guidance for the selection and management of confined disposal areas. It is hoped that the results of the studies in Task 2D, together with interactions with different regulatory agencies, will help in establishing criteria and guidelines on a regional basis for regulating the disposal of dredged material in confined areas.

3. Five research work units were defined within Task 2D:

- a. Work Unit 2D01. Physical and Chemical Characterization of Contaminated Dredged Material Influent, Effluents, and Sediments in Confined Upland Disposal Areas (reported in Hoeppel, Myers, and Engler¹).
- b. Work Unit 2D02. A Study of Leachate from Dredged Material in Upland Disposal Sites and/or in Productive Uses (Mang et al.²).
- c. Work Unit 2D03. Physical and Chemical Monitoring of River Sediments and Water and Confined Disposal Area Dredged Material Influent, Effluents, and Sediments During High-Solids Dredging of a PCB Spill (also reported in Hoeppel, Myers, and Engler¹).
- d. Work Unit 2D04. Characterization of Confined Disposal Area Influent and Effluent Particulate and Petroleum Fractions (Lu et al.³).

e. Work Unit 2D05. Physical and Chemical Characterization of Dredged Material Sediments and Leachates in Confined Land Disposal Areas (Yu et al.⁴).

Work Units 2D01, 2D03, and 2D04 dealt with influent-effluent characterization at active land disposal sites. Work Units 2D02 and 2D05 dealt with potential groundwater impact of inactive sites.

4. Figure 1 shows the approximate locations of the confined disposal sites that were studied; the exact locations and descriptions of the sites are listed in Table 1. Most of the sites were located on the Atlantic and Gulf Coasts of the United States where dredging activities are a necessity for ongoing barge and ship commerce.

5. Physical processes operative in active and inactive sites are illustrated in Figures 2 and 3. Figure 2 depicts a disposal area during active dredging and disposal; the influent dredged material, which consists mostly of dredging site bottom waters, interacts with the disposal area environment (i.e., the atmosphere, vegetation, groundwater, and sediments at the site), and the modified effluent is then usually discharged over a weir structure to surface receiving waters. Figure 3 is a schematic diagram of leachate migration in inactive disposal sites; it also shows the sampling locations for the field study (Work Unit 2D05). Figure 4 describes the divergent groundwater conditions prevalent at the four field sites monitored for leachates. A diagram showing the column setup and sampling scheme for the laboratory lysimeter study (Work Unit 2D02), a simulation of leachate migration from dredged material through an underlying soil profile, is shown in Figure 5.

Studies at Active Sites

6. Due to the dynamic nature of effluent discharge, Work Units 2D01, 2D03, and 2D04 (2D01 and 2D04 Technical Reports) are collectively discussed under "Active Confined Disposal Sites," Part II of this report. The individual studies incorporated into the Active Site Work Units include the following:

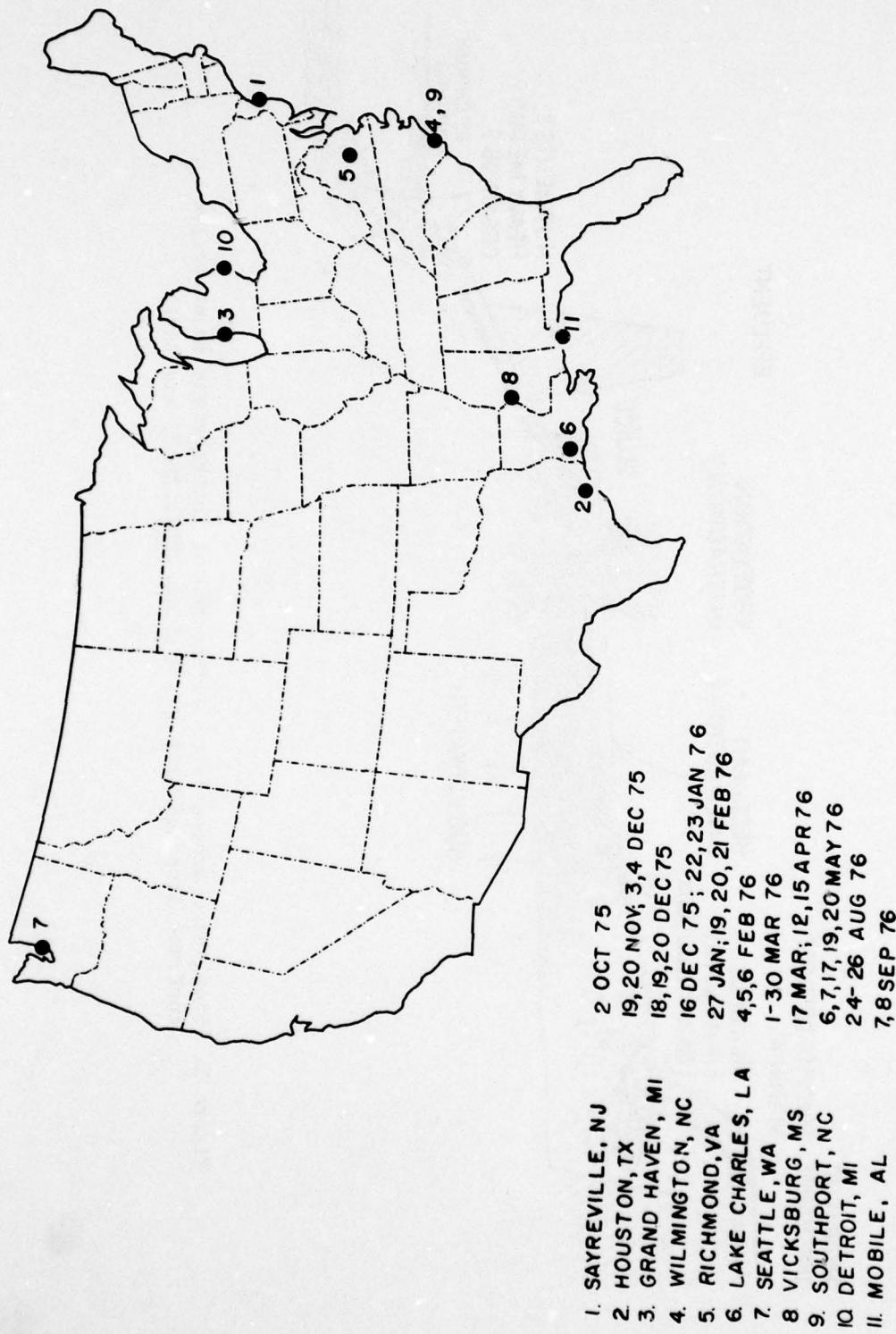


Figure 1. Geographical distribution of the sampled confined disposal sites, with dates of influent-effluent (active site) sampling

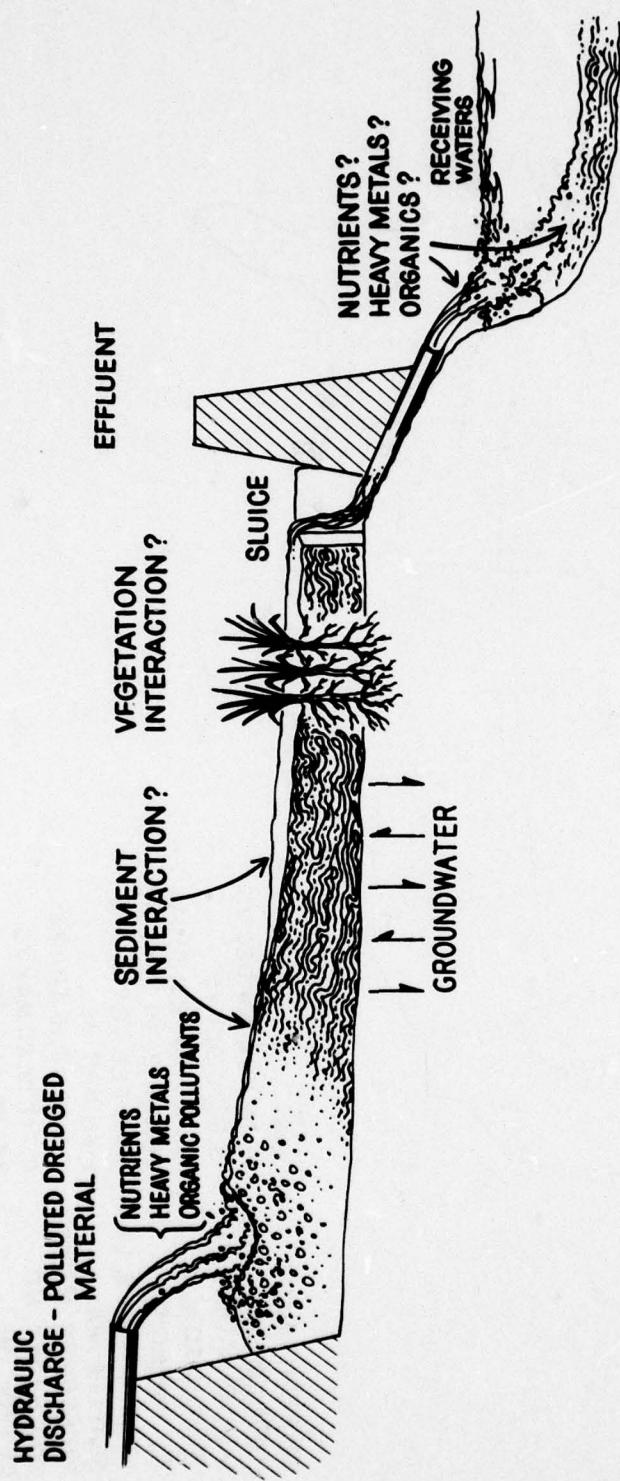


Figure 2. Simplified schematic of environmental interactions in an active confined disposal site (Work Units 2D01, 2D03, and 2D04)

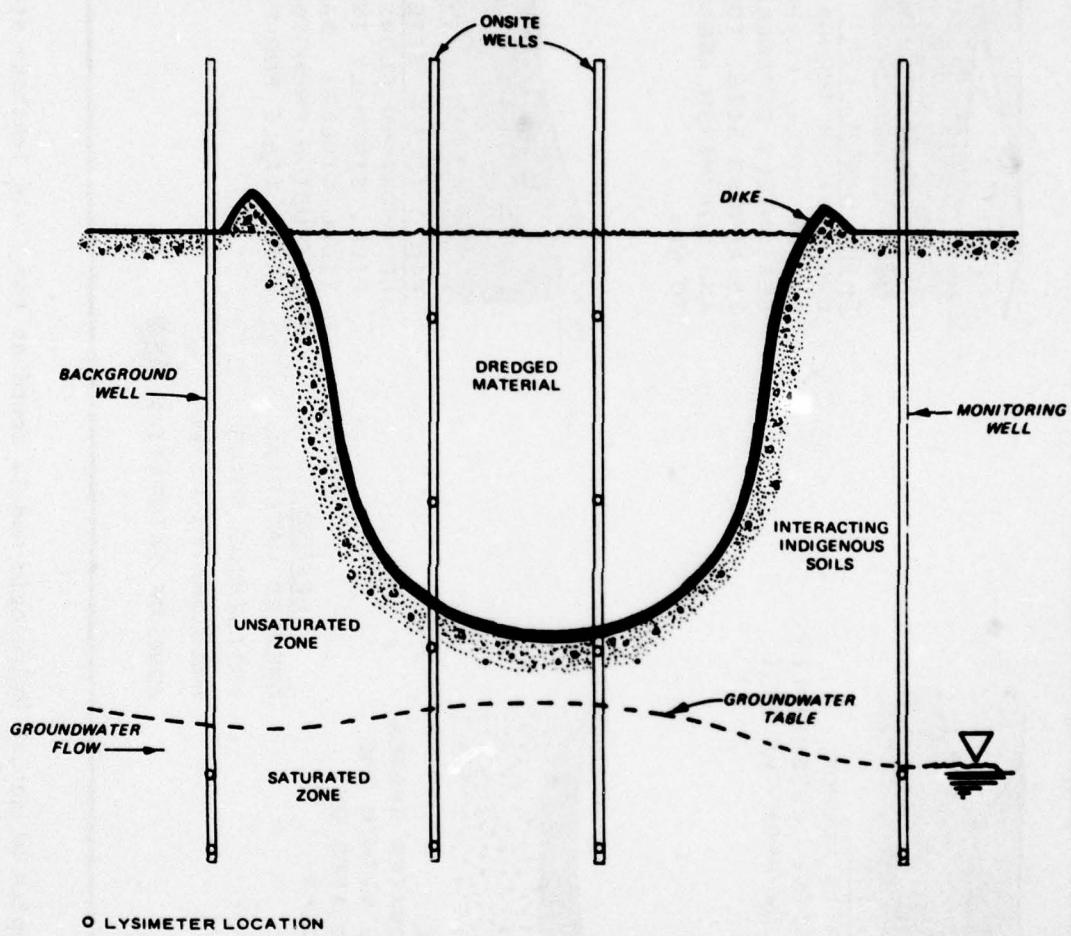


Figure 3. Simplified schematic of leachate sampling in the inactive confined disposal sites (Work Unit 2D05)

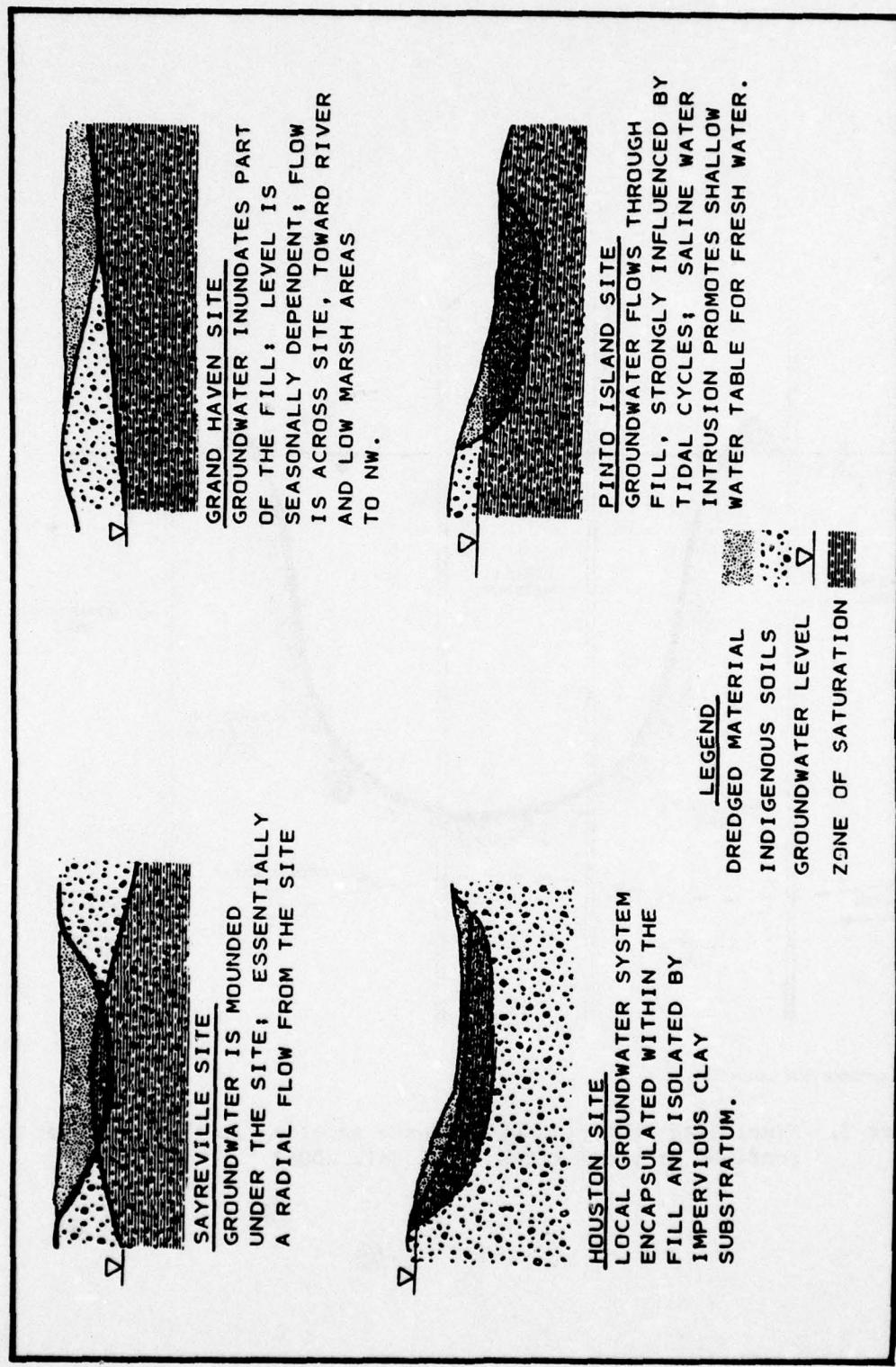
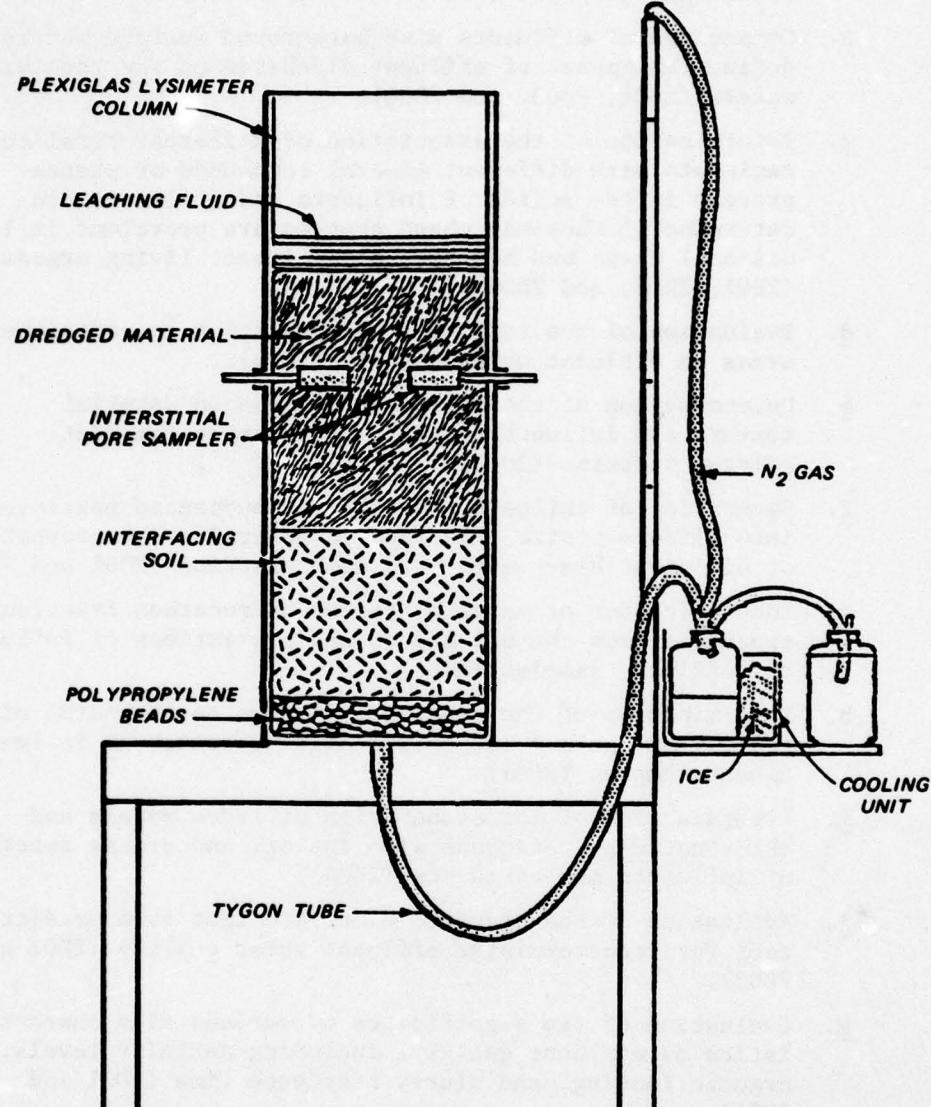


Figure 4. Examples of actual hydrologic models found at the field leachate study sites
(Work Unit 2D05)



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Figure 5. Laboratory simulated leaching study
 (Work Unit 2D02)

- a. Physical and chemical comparisons of influent and effluent samples (11 sites) to help define some of the interactions occurring in land containment areas during active disposal of dredged material (2D01, 2D03, and 2D04).
- b. Comparison of effluents with background surface waters to define the impact of effluent discharge on the receiving waters (2D01, 2D03, and 2D04).
- c. Determination of the association of different metal contaminants with different mineral compounds or phases present in the solids of influents and effluents, to determine if chemical phase changes are prevalent in land disposal areas and how they might impact living organisms (2D01, 2D03, and 2D04).
- d. Evaluation of the influence of vegetation in containment areas on effluent water quality (2D01).
- e. Determination of the influence of dredged material texture and influent solids content on contaminant release patterns (2D01).
- f. Separation of influent and effluent suspended particles into different size fractions to determine the proportion of different heavy metals in each fraction (2D01 and 2D04).
- g. Identification of major petroleum hydrocarbon fractions separated from the oil and grease extractions of influent and effluent samples (2D04).
- h. Determination of the settleability (up to 48 hours) of the oil and grease and chlorinated hydrocarbons in influent samples (2D04).
- i. Determination of the association of trace metals and chlorinated hydrocarbons with the oil and grease fraction of influents and effluents (2D04).
- j. Evaluation of the standard elutriate test as a predictive tool for predetermining effluent water quality (2D01 and 2D03).
- k. Evaluation of the significance of various site characteristics on effluent quality, including salinity levels, organic loading, and slurry residence time (2D01 and 2D04).
- l. Comparison of effluent quality with existing surface water quality regulations (2D04).

Studies Concerning Inactive Sites

7. Work Units 2D02 and 2D05 represent a laboratory scale model simulation and a 9-month field study concerning the potential migration of dredged material leachates through the underlying soil and into groundwaters. Due to the longer period of time required for leachate movement, these work units are discussed under "Inactive Confined Disposal Sites," Part III of this report. The individual studies discussed include the following:

- a. Determination of the chemical composition of subsurface leachates generated from different types of dredged material (2D02 and 2D05).
- b. Comparison of background groundwater quality to down-gradient impacted groundwater (2D02 and 2D05).
- c. Determination of the variability (both horizontally and vertically) in selected physical and chemical parameters for both the natural soil interstitial water and dredged material interstitial water in or near actual disposal sites (2D05).
- d. Evaluation of the effect of varying environmental conditions on subsurface leachate quality (2D02 and 2D05).
- e. Evaluation of the sediment (dredged material and soil) characteristics in relation to the chemical composition of subsurface (dredged material and soil) leachates (2D02 and 2D05).
- f. Determination of the effect of underlying soils on the quality of subsurface leachates generated from dredged material (2D02 and 2D05).
- g. Comparison of subsurface leachate quality with existing water quality regulations (2D02 and 2D05).

8. In the field leachate study, both onsite and offsite (background and monitoring well) water samples were collected at four different depths. It was felt that if migration of contaminants were taking place, gradational concentration changes could be traced in the offsite down-gradient groundwater. The laboratory lysimeter study (2D02) involved five different types of dredged material combined with two different underlying soils in 30-cm-diam Plexiglas columns.

9. While the column simulation study was designed to provide answers to potential problems concerning impact on groundwaters of leachates, this laboratory investigation could only simulate a small number of the environmental variables occurring in nature. Thus, the results can only be used as guidelines for predicting potential effects of disposal within confined sites. Due to the dilution factor that may be present under actual conditions, these laboratory findings should represent worst case simulations. The field study, on the other hand, showed the necessity of monitoring leachates in the confined disposal sites (Table 1) for a long period of time (e.g., greater than 1 year) in an organized and systematic manner with replicate samples. This is necessary in order to understand the spatial and temporal nature of the leachate problem, especially in evaluating long-term effects on receiving waters.

Objectives

10. Considerable attention is being given to confined disposal facilities because of their high initial cost. In many instances the use of large containment areas for a waste disposal-type activity is incompatible with good land use and land management practices. In addition to the problems of surface and groundwater contamination, improperly managed land disposal can be associated with other ill effects such as providing a breeding ground for mosquitos, odor emission, and proliferation of undesirable wildlife. However, many benefits may also be realized from properly planned and managed confined disposal operations, including the production of fertile land, filling of strip mines and flooded quarries, and creation of beneficial wetlands and more diversified habitats for wildlife.

11. This report synthesizes the most significant results of the research performed to date on effluent and leachate quality and control, while also touching on the limitations and problems of confined disposal operations. Finally, soluble and total contaminant levels in effluents and leachates are interfaced with existing criteria (Tables 2 to 4) to

evaluate potential impact. However, this process actually pointed to a need for developing more realistic criteria that are reflective of situations actually encountered in the field; e.g., basing criteria on certain particle size and geochemical fractions which can be shown to be readily assimilated by living organisms.

12. In this parameter-oriented report, every effort has been made to define the phase in which each individual parameter is reported. Considering the variability in sample handling at different research laboratories, a specific identification of the analyzed phase is given; e.g., soluble (<0.05 or <0.45 micron (μ) size) phase and particulate (total or nonfilterable solid) phase. Distinct differences between dredged material from a freshwater versus brackish water source are discussed in terms of both the dredged material composition and established criteria governing the source. Criteria for discharges into brackish and fresh surface waters are given in Table 2; criteria governing leachate water quality for freshwater and marine environments are listed in Tables 3 and 4, respectively.

13. Due to the shortcomings and conflicts in some of the previously acquired data, including the wide variability noted in the discussed work units, future guidelines must be flexible and subject to change as additional information becomes available. In this report, no overall statistical interpretation of data is presented, either because sufficient replicate data were not available or because the statistics have already been discussed in each respective report. In general, the results and discussion are presented from a parameter point of view, emphasizing similarities encountered at each site or in different work units. Where noted differences between sites were deemed interpretable, the mechanisms responsible for the differences are discussed.

14. References on the environmental effects of land disposal of dredged material are very limited. The literature cited in this and the 2D Work Unit reports showed findings which in most cases were in agreement with the results obtained from the work units.

PART II: ACTIVE CONFINED DISPOSAL SITES

General Information

15. Influent, effluent, and background surface water samples from nine confined disposal sites were studied by Hoeppel et al.¹ (2D01). Field measurements included salinity, conductivity, dissolved oxygen, and pH of influents and effluents; Eh and pH of freshly dredged sediments in the sites were also measured. In addition, the following parameters were determined in the laboratory: Coulter counter suspended particulate size determinations, mechanical particle size (percent sand, silt, and clay), total centrifuged solids, nonfilterable (suspended) solids, settleable solids, cation exchange capacity, alkalinity, combined nitrogen (organic, ammonia, nitrate, and nitrite-N), phosphorus (total and orthophosphate-P), total carbon (inorganic and organic-C), chlorinated pesticides, PCB's, oil and grease, sulfides, major ions (calcium, magnesium, potassium, sodium, chloride, and sulfate), and trace metals (iron, manganese, zinc, copper, cadmium, lead, nickel, chromium, mercury, arsenic, vanadium, selenium, and titanium). Similar studies were carried out at two additional sites by Lu et al.³ (2D04) in conjunction with a size fractionation study of influent and effluent suspended particulate matter.

16. In this report, the averages and ranges for parameters have been combined from the 2D01 and 2D04 reports for four freshwater sites and for six brackish water sites.* Individual sites are discussed only when diverse opposing trends are indicated. The most useful comparisons are between influents and effluents to determine disposal area interactions and between effluents and background surface waters to assess the potential impact of the effluents on the receiving waters. Effluent parametric values are also compared with existing and/or proposed water quality

* The brackish water Sayreville, N. J., site was excluded from the calculations since only one sampling was performed and a true influent sample was not obtained for comparative purposes.

standard/regulations to evaluate the potential for water contamination. The experimental results are compared with criteria (presented in Table 2) for the following four water classifications:

- a. Fresh water for public water supply.
- b. Fresh water for aquatic life.
- c. Agricultural water for irrigation.
- d. Marine water for aquatic life.

Within each classification are water quality limits from three entities: the Environmental Protection Agency (EPA), National Academy of Sciences (NAS), and the National Technical Advisory Committee (NTAC). These criteria are considered to be conservative since they are based on total concentrations while much of the particulate constituents may not be immediately available for biological uptake. Also, effluent dilution by receiving waters greatly reduces the concentration levels of discharged pollutants. It must also be kept in mind that it may be necessary to amend the present water quality criteria after additional knowledge is acquired concerning the ecological significance of different soluble and particulate fractions.

Total Solids and Particle Size Distribution

17. Total solids and particle size distribution are important parameters in evaluating the impact of upland dredged material disposal. Solids removal is the most important factor in determining the acceptability of a confined disposal area. The level of suspended solids can be greatly decreased by short-term detention while dissolved solids are not expected to change significantly during normal disposal area detention periods. The significance of this parameter can be summarized as follows:

- a. Suspended solids or slow-settling solids contribute to turbidity.
- b. Suspended solids reduce the penetration of light in the receiving waters, hence affecting photosynthetic activity.
- c. Suspended solids may have a deleterious effect upon aquatic life by placing an oxygen demand on the receiving waters, thus depleting the dissolved oxygen level.

- d. High suspended solids levels can be detrimental to filter-feeding organisms.
- e. Small particles contain large specific surface areas which can serve to adsorb other contaminants and facilitate their transport to the receiving waters.
- f. Slow-settling particulates may cause greater impact on the receiving water quality because of prolonged contact time.

18. The averages and ranges for total solids content and the particle size distribution are presented in Table 5.

Disposal area interactions:

influent-effluent characteristics

19. Nonfilterable (suspended) solids reduction during containment averaged 95 percent for brackish water sites and 99 percent for freshwater sites. Actively growing vegetation in confined disposal areas appeared to be proficient in filtering out suspended solids in the slurries. The low reduction of total or suspended solids in some samples was due to short detention times, resulting either from observed short-circuiting of flow in the disposal area or from overuse of the capacity of the site.

20. The presence of sand in the effluent discharge is indicative of turbulent flow at the weirs, combined with overfilling of a site with sediment. Coulter counter analyses of the suspended solids fraction in influent and effluent slurries and background waters suggest that there was a shift from smaller (mainly clay-sized) particle fractions in the influents to larger diameter (silt-sized) suspended particles in the effluents and background waters. Formation of low-density iron hydrous oxide particles (flocs) during disposal site detention and/or the very slow settling of mainly silt-sized organic particles (e.g., humic substances) may account for this increase in the size of the suspended particulates.⁵

21. The major conclusions concerning containment area solids removal are as follows:

- a. Longer detention times in conjunction with actively growing vegetation in containment areas increase the efficiency of solids removal, resulting in less total pollutant loading in the effluent.

- b. The presence of low-density solids, possibly organic detritus and fine iron oxides, together with very fine clay-sized minerals, may require very long detention times or the use of flocculants to obtain a clear effluent.
- c. Turbulent water conditions near the effluent weir, combined with shallow water depth, can increase the solids content (and thus the levels of total contaminants) in effluents.

Impact on receiving waters:
effluent-background water
characteristics

22. Average levels of nonfilterable (suspended) solids in the brackish and freshwater effluents were 52 and 97 times higher, respectively, than average levels in the background water samples. Due to the fact that most trace contaminants are associated with suspended particles, turbid effluent discharges from confined areas may pose environmental problems. The suspended solids can serve to transport adsorbed inorganic and organic contaminants, as well as modify the physical and chemical environment in the receiving waters near the discharge area. Additionally, the finer (colloidal) particles may readily be ingested by aquatic organisms. However, due to lack of understanding on the availability to aquatic organisms of particulate-associated contaminants, the true biological significance of these particles can only be tentative at this time.

Salinity

23. The suitability of water for many beneficial uses is determined by the level of salinity. Increased levels of salinity in freshwater environments may result in deleterious effects to soils and vegetation as well as impact aquatic life and reduce the utility of receiving waters. Salinity levels at the studied sites during sampling are given in Table 6.

Disposal area interactions:
influent-effluent characteristics

24. The salinity increase during confined disposal averaged 11 percent for brackish water sites and 76 percent for freshwater sites. However, the salinity increase in effluents from the freshwater sites

amounted to only about 0.3 ‰. The observed increases in salinity at most of the specific sites were supported by chloride and soluble sodium values. Evaporation and transpirational loss from vegetation were probably the major mechanisms affecting the increased effluent salinity. No relationship was established between salinity and contaminant mobility. Field observations point to some deleterious impact of highly saline dredged material on vegetation.

25. In general, little correlation could be found between salinity and contaminant mobility (except for the major ions found in seawater). Slight increases in salinity were noted in effluents from most disposal areas, although at a few sites dilution by rainfall or groundwater masked evaporative loss.

Impact on receiving waters:
effluent-background water
characteristics

26. The very low salinity values for freshwater effluents, combined with dilution effects, indicate no adverse impact on the receiving waters. The average salinity of brackish water effluents was approximately two times the average level in the background water samples. It should be noted that the salinity differences between effluents and background surface waters were mainly caused by salinity stratification at the estuarine dredging sites (most of the dredged slurry is bottom water). However, effluent salinity will most likely impact the surface water near the effluent discharge. Regardless of the divergence in salinity values, the water quality impact should most often be localized and short-term and is not considered to be a significant factor; dilution by the receiving waters should reduce the salinity to acceptable levels. However, if salinity stratification is pronounced in the receiving waters, especially in a small isolated body of water, subsurface discharge might be a feasible alternative (provided that temperature, oxygen levels, or other effluent characteristics do not suggest a greater potential impact). Repeated applications of saline influent slurries can cause damage to upland or freshwater lowland vegetation. Salt-tolerant vegetation is recommended for confined disposal areas in brackish water regimes.

factors in governing dissolved oxygen levels in effluents.

30. In general, the average increase of dissolved oxygen in brackish and freshwater effluents during confined disposal area sampling was comparable, although the average freshwater effluent value was higher than that observed for the brackish water effluents. However, influents were also respectively higher at the freshwater sites, probably due to the greater contact of the slurry with air during hopper dredge transport for two of the freshwater disposal operations.

31. It was observed that there were very large fluctuations in dissolved oxygen levels at different depths and time intervals in waters adjacent to the effluent weirs within the confined disposal areas. Subsurface waters were invariably found to be lower in dissolved oxygen than were surface waters. Also, turbulent mixing at the weir was shown to be an important factor influencing dissolved oxygen levels.

32. Consideration should be given to the design of effluent weirs to increase the dissolved oxygen levels without inducing upwelling of oily emulsions and suspended sediment from the bottom of the confined areas.

33. Little significance can be attached to the temperature data (Table 6) since slurry detention in most disposal areas resulted in little temperature change. Only a small net temperature increase was noted at both the brackish and freshwater sites. Temperature variations can affect the concentrations of most chemical constituents in solution.

Impact on receiving waters:
effluent-background water
characteristics

34. The average background water dissolved oxygen levels, based on limited data (Table 6), were 1.7 and 1.2 times the average levels in brackish and freshwater effluents, respectively. The impact of low dissolved oxygen concentrations is primarily dependent on the size, rate of mixing, temperature, and stratification of the body of water receiving the effluent. Assuming an effluent dilution ratio of 5 in the receiving waters, the resulting background water dissolved oxygen

levels would be 7.4 and 9.0 mg/l, respectively, for brackish and freshwater effluents. These concentrations are above the minimum EPA marine water quality criterion of 6.0 mg/l (Table 4). It is important to study dilution factors for discharges from each site so that proper dissolved oxygen levels can be maintained in the receiving waters.

35. The water quality impact of dissolved oxygen levels in effluents on receiving surface waters should be, in most cases, negligible. Efficient management of the disposal area and the resulting dilution by the receiving waters should increase the dissolved oxygen concentrations to acceptable levels.

36. It is recommended that, wherever possible, the intended receiving waters be characterized by a high rate of mixing and that dissolved oxygen stratification in the receiving waters be considered. Perhaps subsurface discharge of some effluents would cause less impact where dissolved oxygen stratification is prevalent.

37. The very small temperature fluctuations noted during containment of dredged material (Table 6) should not impact the receiving waters.

Alkalinity and pH

38. Alkalinity and pH are important parameters since they are indirect indices of conditions which can greatly affect concentrations of most metal species. Changes in pH of a couple of units in a given environment will affect the well-being of most aquatic organisms. Although large differences in alkalinity levels seem to have less impact than pH on most biota, the survival of certain sensitive organisms is dependent on static levels. Alkalinity and pH are inter-dependent.

Disposal area interactions: influent-effluent characteristics

39. The reduction of alkalinity values during confinement averaged 25 percent for brackish water sites and 32 percent for freshwater sites (Table 6). A slight increase of average pH was observed, viz., 0.2 pH

units for brackish water effluents and 0.5 pH units for freshwater effluents (Table 6).

40. The decrease in average effluent alkalinity is in agreement with the overall geochemical phase investigation, which showed an increase of many trace metals in the carbonate phase of effluent solids. The decrease in alkalinity for individual sites can be due to a combination of factors; e.g., the uptake of carbon dioxide during photosynthesis and the subsequent pH increase, which may promote the precipitation of calcium carbonate under certain solution conditions. Also, the decrease in alkalinity may be a result of the neutralization of acid generated by oxidation reactions, especially of ferrous iron and sulfide.

41. Several sites showed an increase in effluent alkalinity during confinement. The increase in alkalinity may be due to microbial degradation of organic carbon to carbon dioxide and organic acids, followed by the dissolution of solid metal carbonate to yield predominantly bicarbonate species. Changes in alkalinity were correlated with field observations, such as the presence or absence of algal blooms in the ponded areas.

42. In conclusion, the decrease of alkalinity during confinement was probably the result of pH increases caused by the uptake of carbon dioxide during algal photosynthetic activity and the subsequent precipitation of calcium carbonate. Following the same reasoning, the increase in alkalinity is probably best interpretable in terms of an increase in microbial degradation of dead vegetation; the released carbon dioxide, although not directly affecting alkalinity, can promote the dissolution of metal carbonates under the increased acidic conditions, yielding generally more soluble bicarbonate species. However, if carbonate solids are lacking in the sediments, then increased acidity will cause a decrease in titratable alkalinity. Alkalinity can also be directly produced by the bacterial decomposition of soluble organic carbon.

43. It was found that high pH values for effluent waters can be promoted by lengthy detention of site surface waters. The resulting low suspended solids levels can accentuate the growth of algae, and algal photosynthesis induces the pH rise. High levels of suspended solids also seemed to buffer the water pH to lower levels while thick vegetation in a site can decrease the light intensity at the sediment or water surface, thus impeding photosynthesis.

Impact on receiving waters:

effluent-background water characteristics

44. The brackish and freshwater effluent alkalinity values (Table 6) were 5.1 and 1.2 times, respectively, the average levels in the background water samples. The EPA, NAS, and NTAC criteria do not prescribe alkalinity limits for marine water quality. Also, no limits have been established for drinking water supplies. In itself, alkalinity is not considered to be detrimental to humans or most aquatic life. Additionally, high background alkalinity levels are generally the rule near most dredging areas. However, highly alkaline waters are usually unpalatable and consumers tend to seek other supplies. It is concluded that the alkalinity of the brackish and freshwater effluents should not have an adverse effect on the receiving waters.

45. The average brackish and freshwater effluent pH values given in Table 6 meet the applicable water quality criteria (Table 2); hence, no water quality impact is indicated. A potential impact does exist if the pH values exceed 8 in the presence of high ammonia levels. Under these conditions, high concentrations of un-ionized ammonia can form, which are very toxic to most aquatic life. Since effluent pH values greater than 8 were rarely observed at the sites in conjunction with high discharge rates, brackish and freshwater effluent pH levels should not have an adverse impact on the receiving waters. However, pH values for background waters were consistently slightly lower than for effluent discharges.

Cation Exchange Capacity (CEC)

46. The significance of cation exchange capacity is not well defined. In general, high cation exchange capacity may present potential long-term problems in releasing trace contaminants from solids under certain conditions.

Disposal area interactions: influent-effluent characteristics

47. The cation exchange capacity levels increased an average of 24 percent for brackish water effluents and 89 percent for freshwater effluents (Table 6). However, one brackish water site showed a reduction of 59 percent.

48. Although mechanical particle size analyses (settling tests) showed an overall increase in the "clay" fraction of effluent suspended solids, direct size analyses (Coulter counter analyses) showed a net shift from clay-sized to silt-sized suspended particles during residence of dredged material in disposal areas. This suggests that low-density flocs were forming (e.g., iron oxide or organic particles), which generally contain very large surface areas for adsorption (exchange) of metal ions because of their porous nature.

49. The confined land disposal of dredged material usually resulted in an increase in cation exchange capacity for the slurry solids. This is directly related to an observed increase of the total exchangeable metals. Also, sediment texture was shown to affect the release of pollutants. Coarse-grained sediments containing moderate organic levels released higher concentrations of most trace metals compared with fine-grained sediments containing an equivalent organic content. Low apparent release of heavy metals from fine-grained sediment may be partly due to adsorption of the released metals by the fine particulate matter.

50. It was observed that total sediment organic content and permeability appeared to be important factors in controlling the mobility of trace metals. This can happen through indirectly regulating pH and changing the oxidation-reduction status (Eh) of the sediment environment or through organo-metallic interactions.

**Impact on receiving waters:
effluent-background water
characteristics**

51. The physical and chemical properties of a dredged material slurry (i.e., Eh, pH, particle size distribution, alkalinity, and salinity) can control the distribution and availability of chemical species. The potential impact of effluent solids with a high cation exchange capacity cannot be overlooked, because potentially toxic trace metals can be transported with the suspended solids and later exchanged in the receiving surface waters. While the extent of redissolution has been shown to be very low, the potential long-term impact on the receiving waters cannot be well defined at present. However, efficient suspended solids removal at a well-managed disposal site should negate the impact of effluent solids having large cation exchange capacities.

52. In summary, effluent solids with high cation exchange capacities may pose a potential long-term impact on the receiving waters. On the other hand, the water quality impact of low-solids effluents on the receiving waters should be negligible.

Sulfides

53. The presence or absence of sulfides in solution or in association with solids reflects the oxidation-reduction condition of the environment. The oxidation-reduction status, in turn, can greatly affect the levels and chemical forms of trace metals and of other inorganic or organic constituents. Sulfide in solution also presents the problem of toxicity to aquatic organisms.

**Disposal area interactions:
influent-effluent characteristics**

54. Total sulfides reduction during containment averaged 91 percent for brackish water effluents and 94 percent for freshwater effluents (Table 7). The removal efficiency of total sulfides correlated fairly well with the suspended solids removal, viz., 95 and 99

percent, respectively, for brackish and freshwater sites. It is quite probable that much of the solid phase decrease was the result of sulfide oxidation. This was strongly suggested by the data from the geochemical phase fractionation study.

55. The sulfur cycle in sediment-water systems is extremely important in regulating the mobility of trace metals. Soluble sulfate ions, which are in high concentration in seawater and in most estuarine environments, can be reduced in sediments to sulfides under anoxic conditions. In freshwater environments, in the presence of low sulfate concentrations, various sulfur-containing organic compounds may become important in sulfide formation. Sulfide oxidation could result in a significant transfer of metals from poorly soluble sulfide solids to other controlling solids having increased solubility.

56. Most of the total sulfide reduction during containment was attributed to total solids removal. Although oxidation of solid phase sulfides appears to be slow, some sulfide oxidation seems evident with long detention. The oxidation of metal sulfides will probably result in increased mobilities for sulfide-bound trace elements.

Impact on receiving waters:
effluent-background water
characteristics

57. The average total sulfide values in effluents from brackish and freshwater sites were 6.3 and 0.6 times, respectively, the average levels in the background waters (Table 7). Marine and freshwater criteria have not been prescribed for total sulfides. The data indicated that the total sulfide levels measured in freshwater effluents were similar to comparable background surface water levels; this implies that sulfides should have negligible impact on the freshwater receiving waters.

58. Since polluted sediments are usually in reduced states, the controlling solids of the in situ sediments are usually reduced solids, i.e., metallic sulfides. Upon resedimentation of the suspended solids in aerobic environments, other solids such as carbonates, hydroxides, oxyhydroxides, and hydrated oxides can be formed. Under oxidizing

conditions, the newly formed controlling solids will generally have increased solubility. Thus high concentrations of total sulfides in brackish water effluents could have an adverse effect on the receiving waters. However, efficient solids removal in the confined disposal areas and adequate dilution by the receiving waters should minimize the water quality impact.

Nitrogen Compounds

59. Compounds of nitrogen are of great importance because of their role as either nutrients or toxicants in the life processes.

Disposal area interactions: influent-effluent characteristics

60. Soluble ammonia nitrogen removal during containment averaged 30 percent for brackish water sites and 54 percent for freshwater sites. Soluble organic nitrogen removal averaged 46 and 34 percent, respectively, for brackish and freshwater effluents (Table 7). The poor removal of ammonia nitrogen seems related to high influent concentrations of ammonia and total organic nitrogen and to the presence of dead or dormant vegetation in several of the disposal areas.

61. Theoretically, in the oxidizing environment in the surface waters of most confined disposal areas, the observed decrease in ammonia nitrogen and organic nitrogen should result in an increase in the nitrate level as a result of nitrification. The observed average increases in the effluent nitrate levels of 90 and 44 percent, respectively, for brackish and freshwater effluents (Table 7) appear to strengthen this assumption. However, nitrogen balance calculations of data in Table 7 readily indicate that either nitrification was not important or that volatile nitrogen gas loss via denitrification must have depleted the nitrate level concurrently with nitrification; de-nitrification could proceed in the anoxic bottom sediments in the disposal areas. Other factors favoring ammonia nitrogen removal include the presence of actively growing vegetation and algae and ion exchange with fine-textured sediments in the sites.

62. The major conclusions concerning containment area ammonia nitrogen, organic nitrogen, and nitrate nitrogen removal are that dredging operations contribute to a rapid release of ammonia in the solution phase. High ammonia levels in influents are not unusual since ammonia is found in interstitial waters of bottom sediments in relatively high concentrations. The rapid conversion of ammonia to nitrate should be possible in confined land disposal areas because oxidizing conditions usually prevail in the surface water. However, the data show only minor increases in nitrate, with little direct relationship observed between residence time and effluent nitrate concentrations. The findings suggest that residence time affects different disposal areas dissimilarly, with the net nitrogen removal dependent on the overall rates of uptake and release, which seem to occur at the same time. It was concluded that it would be desirable to have actively growing thick stands of vegetation, characterized by large root surface areas, in the confined disposal sites. These would facilitate the removal of soluble ammonia and nitrate nitrogen.

Impact on receiving waters:
effluent-background water
characteristics

63. The average nitrate levels of brackish and freshwater effluents were substantially lower than both the background water levels and applicable water quality criteria (Tables 2 and 7). Therefore, no adverse impact on the receiving waters is indicated.

64. The brackish and freshwater average effluent ammonia nitrogen levels of 15 and 9 mg/l, respectively (Table 7), warrant concern, especially if high pH conditions exist in the disposal area or discharge zone which could promote the formation of highly toxic un-ionized ammonia. At a pH value of 8.0, the potential average un-ionized ammonia concentrations in brackish and freshwater effluents would exceed the freshwater and marine water criteria. The required dilution to minimize the impact on the receiving waters must be determined on a site-specific basis.

Phosphorus Compounds

65. The vital importance of phosphorus compounds in water is exemplified by their effects on biological productivity. With proper management, the discharge of phosphate can increase productivity in a beneficial manner. On the other hand, poor management may result in the problem of eutrophication.

Disposal area interactions:

influent-effluent characteristics

66. Total phosphorus reduction during containment averaged 87 percent for brackish water effluents and 98 percent for freshwater effluents (Table 8). These values closely approximate the suspended solids removal, i.e., 95 and 99 percent, respectively, for brackish water and freshwater effluents (Table 5). This indicates an association of total phosphorus mainly with large-sized (settleable) particulates.

67. The data in Table 8 also show that soluble phase ($<0.45\mu$) phosphorus was mainly in the orthophosphate form, i.e., 56 and 80 percent, respectively, for brackish and freshwater effluents. Average soluble phosphorus in brackish and freshwater effluents was 0.55 and 0.11 mg/l, respectively. Soluble phosphorus removal during containment averaged 57 percent for brackish water effluents; freshwater effluents showed a small increase of 36 percent, possibly because of the much lower influent levels. Orthophosphate removal averaged 72 percent for the brackish water sites, while the freshwater sites showed little change in concentration (Table 8). Under aerobic conditions at neutral pH, the iron phosphate solids are poorly soluble and their formation can limit the soluble phosphate level to approximately 0.09 mg/l. Soluble phosphate levels may also be limited by uptake by vegetation and algae and adsorption by clay minerals and ferric hydroxide or calcium carbonate precipitates.

Impact on receiving waters: effluent-background water characteristics

68. The average brackish and freshwater total phosphorus levels in effluents were 75 and 9 times, respectively, the average background water levels (Table 8). The average total phosphorus levels for

brackish and freshwater effluents greatly exceeded the marine and freshwater quality criteria. The average levels in the background waters also exceeded, to a lesser extent, the marine and freshwater criteria (Table 2). High concentrations of phosphates in effluent solids may pose a potential adverse impact on the receiving waters. Inorganic and organic phosphates are subject to hydrolytic breakdown to orthophosphate, which can promote algal blooms and subsequent organic overloading of the receiving waters.

69. The average brackish and freshwater effluent soluble orthophosphate levels were 5.8 and 0.69 times, respectively, the average background water levels (Table 8). The average brackish water effluent orthophosphate phosphorus level was 62 times the NAS marine water quality limit (Table 4). High levels of soluble orthophosphate are immediately available for algal growth which can lead to eutrophication of the receiving waters. The EPA, NAS, and NTAC criteria do not prescribe orthophosphate limits for drinking water, fresh water for aquatic life, and agricultural water.

Organic Carbon

70. The ecological significance of nonspecific parameters such as organic carbon is difficult to define in precise terms. Nevertheless, it is an important parameter for determining the organic loading in water. The organic content of a water body, either in soluble form or in solid particulates, can affect the water chemistry and water use to a great extent.

Disposal area interactions: influent-effluent characteristics

71. Total organic carbon removal during confinement averaged 95 percent for brackish water effluents and 98 percent for freshwater effluents (Table 9). These removal efficiencies approximate the suspended solids removal of 95 and 99 percent, respectively, for brackish and freshwater effluents. This indicates an association of total organic carbon with mainly large-sized particles.

72. Major shifts in alkalinity during confined area disposal seemed to be promoted by biological activity since high influent alkalinity values were noted when the dredged sediments contained a high total organic carbon concentration. These alkalinity increases could, in turn, immobilize trace metal contaminants as carbonate precipitates on settleable solids.

73. Soluble phase organic carbon did not show a significant decrease during land containment (Table 9). Site specific data also indicate that in most instances the soluble carbon remains fairly constant in influent and effluent filtrates. Hence, soluble organic carbon concentrations are not considered to represent a valid indication of oxygen demand in the system. Certain soluble organic compounds can increase the soluble phase levels of many trace metals by forming soluble complexes. However, the association of metals with soluble organics was not critically evaluated.

Impact on receiving waters:
effluent-background water
characteristics

74. The average brackish and freshwater total organic carbon levels in the effluents were 16 and 2 times, respectively, the average background water levels (Table 9). The EPA, NAS, and NTAC criteria do not prescribe a total organic carbon level for either marine or fresh water. However, the release of soluble organic carbon may promote the mobilization of trace metals by the formation of soluble metal-organic complexes. Therefore, the potential water quality impact resulting from the solubilization of organic matter must be considered.

75. The average soluble phase organic carbon concentrations in effluents were generally similar to those in the background waters; for example, the average brackish and freshwater effluent levels were 1.4 and 0.8 times the respective background water levels (Table 9). The oxygen demand exerted by the microbial oxidation of soluble organic carbon should thus have a negligible impact on the receiving waters.

Oil and Grease

76. Oil and grease measured in the studies generally consisted of petroleum ether- or hexane-extractable organic matter. Most of the oil and grease fraction is derived from petroleum discharges, but natural oils, fats, waxes, and some humic substances will also be included. For the contaminated sediments used in these studies, oil and grease levels should correlate fairly well with pollution levels of petroleum hydrocarbons. Straight carbon chain compounds are by far the dominant component of most petroleum products. However, less prominent components include cyclic and aromatic compounds, which are generally more toxic to aquatic life.

Disposal area interactions:

influent-effluent characteristics

77. Total oil and grease decreased by 93 and 99 percent during slurry detention in brackish and freshwater disposal areas, respectively (Table 9). This indicates that oil and grease presents no problems during land disposal operations. However, high levels of petroleum hydrocarbons (representing the majority of the oil and grease) can form low-density layers, which entrain particles and settle with difficulty. Most of the effluent oil and grease recorded was released when these oily layers were stirred up from the bottom of the site by turbulent flow at the weir or when the slurry detention period was too short. Column settling tests indicated that sediments with high levels of oil and grease slowly released oil and grease to the water column over a period of several days. However, the released concentrations later collected at the water surface and accumulated on the column walls; similar removal could occur through contact of the slurry with vegetation and other solid surfaces within the sites. Meager data for soluble phase (centrifuged) oil and grease in influents and effluents indicate that oil and grease has a high affinity for solids. Soluble levels were generally at very low but comparable levels in both influent and effluent samples (Table 9).

78. Oil and grease fractions from influents and effluents at two sites were separated into major components. Generally, the aromatic

component, which is most toxic to organisms, was below detection limits in the effluents. There was some indication that slow microbial degradation of certain straight chain hydrocarbons was occurring during disposal operations. Nevertheless, the alkane (straight chain) hydrocarbons were most abundant in the effluents.

Impact on receiving waters:
effluent-background water
characteristics

79. Total oil and grease levels were 2.7 times higher in the brackish water effluents than in comparable background surface waters; freshwater effluents were similar to the background water in total oil and grease content (Table 9). Soluble oil and grease was at very low levels in most effluent samples. Therefore, on a quantitative basis, the soluble phase should not impact the water quality of receiving waters.

80. Based on settling column testing, less than 1 percent of the total trace metals present in the suspended solids fractions (effluents) from two sites was associated with the oil and grease extracts. Thus effluent oil and grease does not appear to be an important vehicle for trace metal transport to the receiving waters. However, occasionally high discharges of oil and grease were measured in the field, which could impact the receiving waters if allowed to go unchecked. Additional detention time seems to be required to remove high levels of petroleum hydrocarbons and entrained particles.

Chlorinated Hydrocarbons

81. The deleterious effects of chlorinated hydrocarbons in waterways need not be emphasized here. Among all sites studied, DDT and its derivatives as well as the polychlorinated biphenyls (PCB's) constituted more than 99 percent of the chlorinated hydrocarbons in all samples. Therefore, only these two groups are tabulated and discussed.

Disposal area interactions:
influent-effluent characteristics

82. The chlorinated hydrocarbons generally showed excellent removal efficiencies with the exception of brackish water effluent DDE levels (Table 10). In most instances, removal approached or exceeded the suspended solids removal (Table 5), indicating an association with large-grained particles. The average brackish water removal efficiencies were: DDE (23 percent), DDD (89 percent), DDT (\approx 100 percent), and PCB's (96 percent). The average removal efficiencies at freshwater sites were: DDE (98 percent), DDD (99 percent), DDT (96 percent), and PCB's (96 percent).

83. The predominant transformation product of DDT in anaerobic environments appears to be DDD. Its preferential formation, which seems to be microbially induced, is accentuated by both increasing temperature and high levels of organic matter. The main degradation product of DDT in aerobic sediments is DDE. The reaction is catalyzed by certain minerals and heavy metals, especially by iron oxides, and is accelerated by elevated temperature. The conversion to DDE appears to be microbial at lower pH and temperature.⁶ The degradation of PCB's, although slow, is facilitated by many microorganisms,⁷ especially if the PCB's are emulsified in the water phase.⁸

84. Settling tests were performed by Lu et al.³ to determine the fates of oil and grease and chlorinated hydrocarbons and their interactions in the water column after disposal. After 48 hours of resettling, all of the chlorinated hydrocarbons were removed to the following low levels: brackish water column total DDE, DDD, and DDT (99.7 percent) and total PCB's (\approx 100 percent); freshwater column total DDE, DDD, and DDT (99.7 percent) and total PCB's (99.2 percent). Analysis of the suspended oil and grease fraction in the settling test columns indicated that the chlorinated hydrocarbons were not strongly associated with the oil and grease fraction. These tests also indicated that chlorinated hydrocarbon compounds are generally strongly bound with settleable particles.

85. The data show that the chlorinated hydrocarbon compounds generally showed excellent removal efficiencies during disposal area detention, indicating that they should not usually impact receiving waters. However,

DDE was not removed very efficiently at a number of sites during containment, but comparable levels were observed in background water samples. The nearly complete removal of chlorinated hydrocarbons during column settling tests also indicates that their release into the solution phase should be negligible during the usual detention period for confined disposal areas.

Impact on receiving waters:
effluent-background water
characteristics

86. The ratios of average brackish water effluent concentrations to average background surface water values were: DDE (0.6); DDD (>6); DDT (below detection limits); and PCB's (>1.5). The ratios of average freshwater effluent concentrations to average background water values were: DDE (1.8); DDD (1.2); DDT (8.5); and PCB's (>9.3) (Table 10). DDE isomers and PCB's were the most prevalent chlorinated hydrocarbons in dredged material effluents. The results show that DDE isomers at most sites were difficult to remove by land containment.

87. The average freshwater effluent total chlorinated pesticide level was 252,500 times the EPA and NAS criteria for freshwater aquatic life. The average freshwater effluent total PCB level was 465,000 and 930,000 times, respectively, the EPA freshwater aquatic life and NAS drinking water criteria (Table 2). Marine water criteria have not been prescribed for total PCB's and chlorinated pesticides. It is apparent that the effluent total chlorinated pesticide and PCB levels could have an adverse impact on the receiving waters. It should be noted that the applicable water quality criteria for total chlorinated hydrocarbons and PCB's may be unrealistic. Chlorinated pesticides and PCB's have very low solubilities in water, being of the order of less than 0.1 $\mu\text{g/l}$ (ppb). This level is 50 to 100 times the applicable water quality criteria (Table 2). Although only trace quantities of chlorinated hydrocarbons are usually in solution, dissolved organic matter, especially fulvic acids, can significantly increase the solubility of organochlorine compounds.⁹ Availability of soluble chlorinated hydrocarbon species may promote their biomagnification in the aquatic food chain.

Trace Metals

88. Total and soluble trace metal concentrations are important parameters in evaluating impact of upland disposal of dredged material. Trace metals associated with total solids are a very important factor in determining the acceptability of a confined disposal area. The redissolution of solid phase trace metals may pose some problems due to possible biological uptake by aquatic organisms. The averages, ranges, and removal efficiencies for total and soluble trace metals are listed in Table 11.

Disposal area interactions: influent-effluent characteristics

89. The effectiveness of a disposal site in removing suspended and soluble constituents is affected by many complicated factors. The removal of particulates is controlled mainly by the detention time of the containment area and the particle size distribution of resuspended sediments. Generally, most of the trace metals were concentrated in the larger settleable solids of the dredged material, i.e. $>8\mu$. Only a very small portion was found to exist in the solution ($<0.05\mu$) phase (Table 11). Therefore, if the metals were uniformly distributed within the solid phase, the removal efficiency of trace metals associated with the particulates should be close to the removal of the total solids.

90. In most instances, the removal of total metals either approached or exceeded the total solids removal. Trace metals that had significantly lower removal efficiencies than observed for brackish and freshwater suspended solids (i.e., 95 and 99 percent, respectively) include titanium (64 percent) and selenium (39 percent) for average brackish water effluents, and mercury (74 percent) for average freshwater effluents.

91. Several reasons can be given for removal efficiencies higher than the total solids removal.

- a. These metals were associated predominantly with larger sized particulates, which are readily removed during the detention time.
- b. During resedimentation, chemical reactions occurred which promoted precipitation of soluble species.

c. The soluble species were adsorbed by clay minerals and/or hydrated oxides of iron and manganese.

92. For metals that showed lower removal efficiencies than the total solids, the following reasons are suggested:

a. These metals were more concentrated in the smaller particulates and could not be effectively removed during the detention period.

b. During resedimentation, chemical or physical reactions may have altered the original constituents to more soluble species.

93. Land containment resulted in a net decrease of most soluble ($<0.45\mu$) metals (Table 11). Brackish water soluble metal levels, which showed average decreases of less than 50 percent in effluents, include manganese (32 percent), mercury (29 percent), vanadium (28 percent), selenium (21 percent), lead (19 percent), zinc (6 percent), nickel (6 percent), titanium (6 percent), and cadmium (2 percent). Freshwater average effluent soluble metal levels of less than 50 percent include nickel (46 percent), arsenic (25 percent), titanium (21 percent), copper (8 percent), vanadium (5 percent), and mercury (0 percent). Soluble metals which showed increases in average effluents are: copper (5 percent) for brackish water effluents and chromium (25 percent) for freshwater effluents.

94. Under oxidizing conditions, newly formed metallic carbonate, hydroxide, and silicate solids could increase the solubility of most trace metals during detention. However, most soluble ($<0.45\mu$) metal concentrations were reduced in the effluent samples. The following reasons are suggested:

a. The solubility-controlling solids might remain primarily as metallic sulfides instead of being predominantly transformed to carbonates, hydroxides, or silicates due to short detention times. Therefore, the concentrations of soluble metals would not be increased because of major geochemical solid phase changes.

b. A decrease of metal-organic complexes in the effluents or adsorption of soluble metals by settleable organic matter may account for the decrease.

c. The soluble iron and manganese concentrations were quite high in the influents; these could be oxidized in the presence of oxygen to form hydrated oxides which could

scavenge (adsorb) most of the other soluble metals from the solution. The increase in cation exchange capacity of effluent solids also suggests the importance of surface adsorption phenomena.

95. The major findings concerning containment area trace metal removal are that the total metals generally showed excellent removal efficiencies; in most instances, removal approached or exceeded the total solids removal, indicating an association with larger particulates. The geochemical transformation data suggest that the concentration of most soluble metals under oxidizing conditions should increase during confinement area disposal, since increases in exchangeable and carbonate phase metals were indicated. The observed general decrease in most soluble metals is the net effect of complex physical (e.g., solids adsorption) and chemical changes occurring during confinement.

**Impact on receiving waters:
effluent-background water
characteristics**

96. The data in Table 11 show that the average brackish and freshwater total metal concentrations in effluents were, in all instances, greater than the average background water levels. The effluent-background water concentration ratios ranged from 1.4 (nickel) to 141 (titanium) for brackish water effluents and from 4.4 (copper) to 80 (arsenic) for freshwater effluents.

97. The data also show that in most instances the average brackish and freshwater effluent total metal concentrations were significantly higher than the marine and freshwater requirements (Table 2). Table 12 gives the effluent concentration to criteria ratios for each metal in both brackish and freshwater regimes. The effluent/criterion factors are based on the lowest of the EPA, NAS, and NTAC marine and freshwater criteria.

98. High concentrations of effluent solid phase metals pose a potential adverse impact on the receiving waters. While the extent of redissolution may be very small, trace metals associated with suspended particulates, including macromolecular organic complexes, may pose some problems due to possible biological uptake and subsequent biomagnification in the aquatic food chain.

99. The data in Table 11 show that the average soluble phase trace metal concentrations were generally similar to the background water levels. The average brackish water and freshwater effluent levels were in the ppb or sub-ppb range for most trace metals. All of the soluble trace metal concentrations were much lower than the marine and freshwater criteria (Table 2); therefore, the potential water quality impact on the receiving waters is considered to be negligible. The effluent/criterion factors for soluble metals are given in Table 12.

PART III: INACTIVE CONFINED DISPOSAL SITES

Laboratory Column Study

General trends

100. Through laboratory experimentation, it was found that the potential impact on water quality from leachates that originate from upland dredged material disposal may be a function of the chemical characteristics of the dredged material and underlying site soils, as well as the planned use for the receiving waters. The laboratory simulation did not incorporate any site-specific factors such as potential dilution effects, climatic variation, and site geology. In general, this study showed that levels of orthophosphate, pH, cadmium, copper, mercury, and soluble lead in leachates from upland dredged material disposal areas should not present serious water quality problems.

101. However, it was found that the levels of ammonia nitrogen, nitrate nitrogen, alkalinity, iron, manganese, total lead, and possibly zinc in leachates might exceed criteria for one of the following: fresh water for public water supply, fresh water for aquatic life, agricultural water for irrigation, marine water for aquatic life, or groundwater. In certain cases, protection of groundwaters prior to the disposal of dredged material to upland areas may be necessary.

Dredged material leachate

102. In the laboratory simulation, dredged material leachate was defined as that fluid that was withdrawn from the interstitial pore water of the dredged material in the lysimeter columns. This fluid comprised the input solution to the interfacing (underlying) soils. The levels of the parameters analyzed for in this phase of the study are of obvious importance because they define the extent of constituent mobilization from dredged material in a simulated upland environment. The basic scientific mechanisms that control constituent mobilization from dredged material are also important, i.e., when a "well-defined understanding" of basic chemical interactions is achieved, it may be

possible to locate or design upland dredged material disposal sites so that the mobilization of potential water column contaminants is minimized.

103. Total organic carbon. The results of this study show that the soluble total organic carbon levels in the interstitial water of the dredged material generally increased throughout the experiment, from initial values of approximately 100 mg/l to final values that ranged from 300 to 600 mg/l in the various lysimeter columns. Due to the complexity of the sediment-water system, the release mechanisms for total organic carbon from dredged material were not readily definable. However, the major controlling mechanism should be hydrolytic splitting of large molecules with resultant biodegradation of the smaller fragments. Nevertheless, not all soluble organic matter (soluble total organic carbon) is readily degradable by microorganisms. No qualitative assessment of the total organic carbon fraction was performed.

104. Alkalinity. Levels of alkalinity gradually increased in the interstitial water throughout the experimentation. Initial values in the various lysimeter columns ranged from 300 to 1100 mg/l; final values ranged from 500 to 1500 mg/l. By comparing the columns, it was found that different columns containing the same dredged material would maintain virtually the same level of alkalinity throughout the experimental period. There were no definitive relationships found to exist between alkalinity and dredged material characteristics such as total organic carbon or calcium. It was speculated that alkalinity levels in dredged material interstitial waters were controlled by the dissolution of carbonate solids due to acid production and metal transformation.

105. Nitrogen compounds. In general, the concentrations of ammonia-nitrogen and total Kjeldahl (organic and ammonia) nitrogen in the interstitial waters of the dredged material were found to decrease throughout the course of the experiment. In certain cases, ammonia-nitrogen levels increased slightly toward the end of the experimental time frame. Final values ranged from 0.11 to 6.7 mg/l for ammonia-nitrogen in the various laboratory columns and from 7.6 to 19.2 mg/l

for Kjeldahl nitrogen. It was ascertained that ammonia nitrogen levels were controlled by the oxidation-reduction state, dilution, and the reaction flux (total nitrogen in the dredged material). It was found that oxidized sediments posed a less favorable environment than reduced (anoxic) sediments for the accumulation of ammonia nitrogen in the interstitial water. Along these same lines, it was hypothesized that the oxidation-reduction state, bioconversion, and dilution controlled the levels of Kjeldahl nitrogen in the dredged material interstitial waters studied.

106. Phosphorus compounds. Both orthophosphate and total phosphorus were quite low in the interstitial waters. At the conclusion of the experiment, the levels of soluble phosphorus approached 0.01 mg/l. It was found that the levels of phosphorus compounds were mainly controlled by the solubility of calcium phosphate solids.

107. pH. After one month of experimentation, pH values in the dredged material interstitial waters approached 8 from initial values of about 6.7. Throughout the course of the experiment, the hydrogen ion concentrations were decreased by two orders of magnitude over initial levels. This decrease was probably caused primarily by less bioactivity, dilution effects, and dissolution of calcite.

108. Major ions. The levels of most of the major ions in dredged material interstitial waters were controlled mainly by the dilution effect or the solubility effect. A summary for each ion studied is as follows:

a. Sodium.

- (1) Experimental phenomena. Sodium in the interstitial water was found to decrease gradually throughout the experiment. Levels in interstitial water from saline dredged material ranged from 6000 to 9000 mg/l initially to from 3000 to 7200 mg/l at the end of the experiment. Levels from freshwater dredged material varied from 70 to 82 mg/l at the outset to 31 to 56 mg/l at the end of the experiment.
- (2) Controlling mechanisms. The study concluded that the dilution effect (by the rainfall leaching solution) was responsible for controlling sodium levels in dredged material interstitial waters.

b. Potassium.

- (1) Experimental phenomena. Potassium in the interstitial water decreased gradually throughout the experiment; initial values ranged from 30 to 600 mg/l, while at the end of experimentation, values ranged from 6 to 300 mg/l in the various lysimeters.
- (2) Controlling mechanisms. The laboratory study concluded that the dilution effect from the rainwater leach was responsible for the decreasing potassium concentrations in the interstitial waters of the dredged material.

c. Calcium.

- (1) Experimental phenomena. The calcium levels in interstitial water changed slowly, increasing slightly during the initial leaching period and decreasing throughout the remainder of the experiment. Values at the end of the experiment ranged from 60 to 220 mg/l in the various lysimeter columns.
- (2) Controlling mechanisms. It was concluded that the solubility effect (controlled by calcite dissolution and precipitation) was the primary regulating mechanism for calcium concentrations in interstitial waters.

d. Magnesium.

- (1) Experimental phenomena. For the saline or brackish water dredged material, the magnesium levels exhibited either no significant changes throughout the experiment or increased only slightly, then decreased again after 1 to 2 months of the rainwater leaching test. Final experimental values ranged from approximately 280 to 770 mg/l. In freshwater dredged material, the levels of magnesium continually decreased throughout the experiment. Final levels ranged from 3.3 to 5.2 mg/l in the various lysimeter columns.
- (2) Controlling mechanisms. The dilution effect was responsible for regulating magnesium levels.

e. Chloride.

- (1) Experimental phenomena. In saline dredged material, the soluble chloride values changed only slightly (either increased or decreased) during the leaching test. In the various lysimeter columns, the final values ranged from 8200 to 13,000 mg/l. At the end of the experiment, in freshwater dredged material, the soluble chloride values decreased about 50 to 65 percent over the initial values. Final values

in the various experimental lysimeter columns ranged from 48 to 68 mg/l.

- (2) Controlling mechanisms. The laboratory study concluded that dilution and ion exchange were important in regulating chloride levels in interstitial waters from saline dredged material. It was also concluded that dilution controlled chloride levels in the freshwater dredged material.

109. Trace metals. Most of the trace metals in the interstitial waters of the dredged material were primarily controlled by complexation or solids adsorption. Generally speaking, the adsorption of trace metals onto the surfaces of solids can account for lower trace metal levels than concentrations which should be released from even the most insoluble chemical compounds. In the experimental systems, it was hypothesized that the most important adsorbents were hydrated oxides of iron and manganese as well as various clay minerals. A summary for each trace metal studied is as follows:

a. Iron.

- (1) Experimental phenomena. Iron in interstitial water exhibited a decrease-increase-decrease cycle. The final levels of iron ranged from about 0.5 to 0.8 mg/l for the saltwater dredged material and decreased to nil for the freshwater dredged material.
- (2) Controlling mechanisms. The laboratory study concluded that experimental trends were controlled by solid transformation (formation of ferric hydroxide and oxyhydroxide solids), organic complexation effects, and oxidation-reduction fluctuations. It was speculated that iron concentration levels were controlled by the formation of iron-organic complexes.

b. Manganese.

- (1) Experimental phenomena. In general, interstitial water manganese levels increased in the initial period of leaching (within two months) and decreased throughout the remainder of the experiment. Values at the end of the experiment ranged from 0.2 to 3.1 mg/l in the various lysimeter columns.
- (2) Controlling mechanisms. It was concluded that experimental trends were controlled by solid transformation (conversion from manganese sulfide to manganese carbonate to oxyhydroxide or variable

component oxide/hydroxides) and oxidation-reduction changes. The laboratory study hypothesized that iron concentration levels were controlled by complexation (formation of soluble chloride and bicarbonate complexes).

c. Cadmium.

- (1) Experimental phenomena. In general, cadmium in the dredged material interstitial water was found to increase in the initial experimental period (2 to 3 months) then decrease throughout the remainder of the experiment. Initial experimental values ranged from 0.6 to 165 $\mu\text{g/l}$; final values ranged from nil to 67 $\mu\text{g/l}$
- (2) Controlling mechanisms. The laboratory study concluded that experimental trends were controlled by changes in the solubility of the controlling solids, from poorly soluble sulfide solids to more soluble carbonate solids. The later decrease may have been caused by adsorption on hydrated oxides of iron and manganese. It was concluded that concentration levels of cadmium were controlled by the formation of soluble cadmium-organic and cadmium-chloride complexes and free ionic cadmium.

d. Copper.

- (1) Experimental phenomena. Soluble copper levels decreased significantly after about 3 months of the experiment. The final concentration of copper in the interstitial water after 6 months was less than 3 $\mu\text{g/l}$ (originally 10 to 70 $\mu\text{g/l}$).
- (2) Controlling mechanisms. It was hypothesized that experimental trends were controlled by solid transformation from poorly soluble copper sulfide to more soluble hydroxy carbonate (malachite), by adsorption onto hydrated oxides of iron and manganese, and by the dilution effect. The study concluded that concentration levels were controlled by soluble complex formation (copper carbonate, copper borate, and/or copper-organic complexes).

e. Mercury.

- (1) Experimental phenomena. Soluble mercury levels in the interstitial waters were maintained at a constant level after 1 month of experimentation (concentrations were usually less than 1 $\mu\text{g/l}$).
- (2) Controlling mechanisms. The laboratory lysimeter study found that experimental trends were initially

controlled by the low solubility of mercury sulfide solids. Throughout the rest of the experiment, adsorption by hydrated oxides of iron and manganese controlled mercury trends.

f. Lead.

- (1) Experimental phenomena. The lead levels in interstitial waters of saline dredged material generally increased to the levels of 250 to 300 $\mu\text{g/l}$ between the beginning and end of experimentation. In freshwater dredged material, the soluble lead generally decreased from about 30 $\mu\text{g/l}$ to below 1 $\mu\text{g/l}$ throughout the course of the experiment.
- (2) Controlling mechanisms. It was concluded that for the saline dredged material, the initial concentration trends were controlled by transformation from less soluble sulfide to more soluble lead carbonate solids; lead organic complexes can also probably account for the maximum soluble levels observed in the experiment. Low final levels were thought to be caused by lead adsorption onto hydrated iron and manganese oxides. For the freshwater dredged material, the low concentration levels of lead could be controlled by solids adsorption.

g. Zinc.

- (1) Experimental phenomena. There were two transport trends observed for zinc in the interstitial waters of dredged material: (a) zinc continually decreased as the experiment progressed, and (b) zinc suddenly increased and then decreased again. The final levels of zinc were in the range of 1 to 20 $\mu\text{g/l}$ (6 months of experimentation) for the various lysimeter columns.
- (2) Controlling mechanisms. Based on equilibrium calculations using the experimental data, the observed zinc levels appeared to be controlled by the solubility of zinc silicate minerals (e.g. willemite). Low levels were probably controlled by adsorption on clay minerals. It was also speculated that zinc concentration levels were controlled by zinc-carbonate, zinc-hydroxide, and zinc-organic complexes. Several mixed systems seemed to be simultaneously involved.

110. Chlorinated hydrocarbons. In the laboratory simulation of dredged material disposal to upland areas, the experimental configuration did not allow for an extensive number of analyses for chlorinated

hydrocarbons. However, in those samples analyzed, the levels of chlorinated hydrocarbons were extremely low, i.e., below detection limits. It was speculated that the reason for these low levels was adsorption by dredged material particles (especially clay minerals, iron and manganese hydrated oxides, and immobile organic matter).

Interfacing soil leachate

111. In the laboratory simulation, the interfacing soil leachate was defined as that fluid that had an opportunity to migrate through both the dredged material and interfacing soil. It most closely simulates the field site leachates that may reach underlying groundwaters. It is for this reason that data from this section of the experiment are compared to existing water quality criteria (Tables 3 and 4). Analyzed levels of constituents in this segment of the laboratory simulation are obviously important in that they define the extent of potential water quality impact. The basic scientific mechanisms that control migration of constituents between soils and dredged material leachates are also important. Once constituent behavior in various dredged material and soil regimes is understood, then upland dredged material disposal sites may be judiciously selected and designed to minimize the potential for groundwater degradation.

112. It was found in this section of the study that the major mechanisms which could affect the attenuation (decreased mobilization) of constituents by the experimental interfacing soils were usually adsorption, precipitation, and ion exchange. However, the elution of highly soluble constituents (e.g., major ions) was usually regulated by dissolution, complexation, desorption, hydrolytic action, ion exchange, and bioconversion reactions. Physical scouring was the principal cause for the release of particulate forms of the constituents from the soil.

113. Total organic carbon. Results from the control sets (interfacing soil with no overlying dredged material) show that organic matter could be leached from the interfacing soils in appreciable quantities. Levels reached as high as 900 and 250 mg/l, respectively,

for the two experimental interfacing soils. The soil that contained a fourfold greater level of total organic carbon also produced the highest levels of soluble total organic carbon when leached with distilled water.

114. In the columns containing dredged material in combination with the interfacing soil with the higher total organic carbon level, the soil did not immobilize significant soluble total organic carbon generated from the dredged material. The columns containing dredged material in combination with the soil having lower total organic carbon showed a release of soluble total organic carbon from the soil to the final leachate. Due to the lack of knowledge concerning organic species and bioreactions in soil, an accurate evaluation of the controlling mechanisms was found to be very difficult.

115. Alkalinity. The results of the lysimeter column experiment demonstrated that alkalinity levels in the leaching fluids could be significantly altered by the interfacing soils. If alkalinity in the influent solution (dredged material interstitial water) was higher than in the leachates from the control soil column, the concentration decreased during percolation through the soil. If alkalinity in the influent solution was lower than in the leachates from the control soil column, the final leachate concentration increased. In any event, final alkalinity levels in the leaching solutions that passed through both a dredged material and interfacing soil ranged from 280 to 1200 mg/l in the various lysimeter columns.

116. As was demonstrated by calculation, alkalinity levels were due mainly to carbonate species. The carbonates of major elements such as calcium and magnesium, and more specifically calcite and dolomite, were probably the primary solids involved.

117. Nitrogen compounds. This experiment found that soils could contribute ammonia nitrogen to leachates during the initial leaching period (<3 months). In most cases ammonia nitrogen was attenuated by soil after this initial period, probably through ion exchange or adsorption. In certain cases where ammonia nitrogen levels increased, the levels might be due to the saturation of adsorption sites or the conversion of organic nitrogen to

ammonia nitrogen. In any event, final experimental ammonia nitrogen values ranged from 1.39 to 7.56 mg/l in the soil leachates from the various lysimeter columns.

118. It was found that, after passage through the experimental interfacing soils, the total Kjeldahl (organic and ammonia) nitrogen levels in the final leachates could be increased significantly over levels in the dredged material interstitial water. Bioconversion of the insoluble complex organic matter into soluble species and the leaching of smaller sized organic matter from the soils are probably the most important mechanisms. Levels of total Kjeldahl nitrogen in the final experimental soil leachate samples ranged from 29.4 to 56 mg/l in the various lysimeter columns.

119. Phosphorus compounds. When contacted by the leaching fluids, the soils released phosphorus compounds during the initial leaching period (<3 months). However, the amount of total phosphorus and soluble orthophosphate phosphorus in the soil leachates decreased to very low levels (<0.01 mg/l) throughout the course of experimentation. It was speculated that phosphate might be initially released from the saturated soils, due to the reduction and dissolution of ferric phosphate solids or iron and manganese hydrated oxide precipitates by the reduced dredged material leachates. However, released phosphate may have later been scavenged by the high levels of calcium in the soil solutions and transformed from iron phosphate to calcium phosphate, which is not directly affected by reduced conditions. These phenomena could explain why the release of phosphate only occurred in the early stages of the experiment.

120. pH. After passage of the leachates through the interfacing soils, the pH generally decreased from the levels measured in the interstitial water of the overlying dredged material. Levels of pH in the final soil leachate samples ranged from 6.95 to 7.65 in the various experimental lysimeter columns. It was suggested that the final pH of the soil leachates would be gradually regulated by carbonate solids in the system. Due to the abundance of calcite, it was speculated that the controlling solid for pH was probably calcite.

121. Major ions. The laboratory lysimeter study showed that many of the major ions could be eluted from soils when contacted by the input leaching solutions. The exceptions to this were magnesium, which was strongly attenuated by the soils, and chloride, which was attenuated or eluted depending upon the soluble chloride levels generated in the dredged material interstitial water (soil system influent). In many cases, it was ascertained that ion exchange was the primary mechanism which controlled both the experimental trends and analyzed levels of the major ions in the soil leachates. A summary of each of the major ions studied is as follows:

a. Sodium.

- (1) Experimental phenomena. It was found that sodium was eluted from one of the experimental soils while being attenuated by the other. This was probably due to the high initial sodium values in the one interfacing soil. For leaching fluids that contained high sodium levels, the attenuative capacity of the interfacing soils decreased as the experimental time frame progressed. Levels of sodium in the final soil leachate samples ranged from 21 to 157 mg/l in those lysimeter columns containing freshwater dredged material and 636 to 5680 mg/l in those containing saline dredged material.
- (2) Controlling mechanisms. This study demonstrated that attenuation of sodium was controlled by the ion exchange mechanism (exchange for calcium ions). Elution was controlled by dissolution of simple sodium solids (salts) from the soil and ion exchange on solids.

b. Potassium.

- (1) Experimental phenomena. Potassium could be eluted from the soils. There was no overall mass removal of potassium by soils. Potassium levels in final soil leachate samples ranged from 24 to 152 mg/l in the various lysimeter columns.
- (2) Controlling mechanisms. The laboratory study concluded that dissolution of simple potassium salts from initially dry soils and ion exchange reactions controlled potassium behavior in interfacing soil leachates.

c. Calcium.

- (1) Experimental phenomena. Calcium was strongly released from the soils. The rate of release was higher in saline dredged material/soil tests than that in freshwater dredged material/soil tests. Calcium levels in the final soil leachate samples ranged from 204 to 306 mg/l for those lysimeters containing freshwater dredged material and from 934 to 4290 mg/l for those containing brackish water dredged material.
- (2) Controlling mechanisms. One conclusion from this laboratory study was that the major source of calcium was ion exchange on soil particles with soluble sodium and magnesium ions. It was speculated that concentration levels of calcium were gradually regulated by calcite (CaCO_3) dissolution.

d. Magnesium.

- (1) Experimental phenomena. Magnesium was strongly attenuated by the soils. The attenuation capacity of the soils decreased as the experimental time frame progressed. Levels of magnesium in the final soil leachate samples ranged from 3.3 to 131 mg/l in the lysimeter columns containing the freshwater dredged material and from 160 to 775 mg/l in those containing saline or brackish water dredged material.
- (2) Controlling mechanisms. This study concluded that concentration levels of magnesium were controlled by ion exchange (replacement of calcium ions on the clay minerals) and the experimental trends were regulated by the availability of exchange sites.

e. Chloride.

- (1) Experimental phenomena. Either attenuation or elution occurred depending on the relative levels of chloride in the leaching solution from the dredged material (input to the soil). Chloride levels ranged from 185 to 260 mg/l in the final soil leachate samples from lysimeter columns containing freshwater dredged material and from 3400 to 9600 mg/l in samples from columns containing saline dredged material.
- (2) Controlling mechanism. It was demonstrated that attenuation of chloride could be controlled by ion exchange and dilution. Elution of chloride was controlled by ion exchange and dissolution.

122. Trace metals. Most of the trace metals in final leachates, that resulted from the passage of leaching fluids through both the dredged material and interfacing soils, were controlled to a large extent by adsorption, dissolution, and/or complexation (depending upon the particular trace metal in question). Adsorption was an important mechanism for the removal of cadmium, copper, mercury, and lead from the soil solution. Dissolution of metallic solids was probably involved when transformation of solids occurred; iron and manganese were prime examples of this effect. Complexation was probably important for accentuating the release of trace metals into the soil leachates. The organometallic complexes are believed to be the most significant species in this regard. A summary for each trace metal studied is as follows:

a. Iron.

- (1) Experimental phenomena. Soluble iron was released from the soils. The amount released was related to the soluble total organic carbon levels in the soil leachates and the cation exchange capacity and organic content of the soils. Final experimental values for soluble iron ranged from 0.2 to 35 mg/l. A large quantity of filterable iron colloids were also released from the soils. Total iron values ranged from 30 to 115 mg/l in the final soil leachate samples from the various lysimeter columns.
- (2) Controlling mechanisms. It was demonstrated that iron levels were mainly controlled by complexation (iron-organic complexes). It was also found that Eh (oxidation-reduction potential) and pH fluctuations and ion exchange also could affect the release of iron. The movement of iron colloids was controlled by physical scouring effect.

b. Manganese.

- (1) Experimental phenomena. The amount of soluble manganese released from the soil containing high initial total organic carbon (1.95 percent) was significant. Final experimental levels ranged from 20 to 120 mg/l in the various lysimeter columns containing this soil. The migration of manganese was low in the soil containing an initially low total organic carbon level (0.55 percent). The levels of manganese in the final leachate samples from columns that contained this soil ranged from 1.6 to 18.5 mg/l.

(2) Controlling mechanisms. The laboratory study concluded that manganese levels were controlled by solubilization, e.g., the solubility-controlling solid(s) were transformed gradually from oxides to carbonate and sulfides under the more reduced conditions in the saturated soils. Complexation (formation of soluble chloride and organic complexes) also seemed to be important.

c. Cadmium.

(1) Experimental phenomena. Cadmium was eluted from the soil blank columns by distilled water. Values of cadmium in leachates from these two control lysimeters were 1 and 4.5 mg/l at the end of the experiment. In general, soluble cadmium in the dredged material leachates was removed by the soils. Cadmium levels in the final column leachate samples ranged from below detection limits to 27 mg/l.

(2) Controlling mechanisms. This study concluded that the trends for cadmium in the control soil sets were controlled by dissolution of cadmium carbonate solids and formation of cadmium-inorganic complexes, mainly soluble chloride complexes. Cadmium levels in leachates from columns containing both dredged material and soil were controlled by precipitation (the gradual transformation of cadmium from carbonate to sulfide solids under more reduced conditions in the soils), adsorption, and decomplexation effects. It was also concluded that cadmium concentrations were regulated by the formation of soluble cadmium-organic complexes.

d. Copper.

(1) Experimental phenomena. Soluble copper could be released in the range of 1 to 5 $\mu\text{g}/\ell$ from soils leached with distilled water (soil control lysimeters). Soluble copper could be attenuated by soils if the influent levels (from the dredged material) were greater than 5 $\mu\text{g}/\ell$, which usually occurred during the initial experimental phase. Final experimental values for soluble copper ranged from nil to 20 $\mu\text{g}/\ell$ in the various lysimeter columns. Significant quantities of particulate-bound copper were leached out only during the initial period of the experiment.

(2) Controlling mechanisms. In the control soil columns it was found that pH (low pH favors dissolution) and complexation effects would control copper levels. The study concluded that pH, Eh, complexation (with

insoluble organic matter), precipitation (sulfide solids), and adsorption (clay minerals, iron and manganese colloids) could regulate copper levels. Physical scouring controlled levels of particulate-bound copper.

e. Mercury.

- (1) Experimental phenomena. There was little change in the soluble mercury levels before and after passage of leachates through the soils. However, particulate mercury was found to be released into the final leachates. Levels for total mercury ranged from 3.5 to 23.7 $\mu\text{g/l}$ in the final soil leachate samples.
- (2) Controlling mechanisms. The laboratory study concluded that low soluble mercury levels were due to attainment of equilibrium conditions in the systems between solubility, adsorption, and complexation effects. Levels of particulate mercury were controlled by physical scouring.

f. Lead.

- (1) Experimental phenomena. Lead was removed by the soils. The attenuation capacity of the soils for lead gradually decreased as the leaching time increased. Values for soluble lead in the final leachate samples from the various columns ranged from below the detection limit to 45 $\mu\text{g/l}$.
- (2) Controlling mechanisms. It was concluded that adsorption (clay minerals, iron and manganese colloids) and solubilization (due to sulfide solid formation) controlled lead to extremely low levels.

g. Zinc.

- (1) Experimental phenomena. Zinc was usually eluted from the interfacing soils. Soluble zinc levels in the final soil leachate samples ranged from 25 to 270 $\mu\text{g/l}$.
- (2) Controlling mechanisms. The laboratory leachate study concluded that the occasionally high zinc concentrations were controlled mainly by zinc-organic complexation, coupled with relatively low adsorption and sulfide formation capacities.

123. Chlorinated hydrocarbons. In general, the individual species of chlorinated pesticides and PCB's were at very low levels, usually below the detection limits of the techniques utilized. Two conclusions were drawn from the limited experimental results:

- a. The levels of soluble chlorinated hydrocarbons in final leachates were usually very low, at levels of less than 1 $\mu\text{g/l}$.
- b. There was also no significant migration of total chlorinated hydrocarbons with the leachates.

As in the dredged material profiles, the chlorinated hydrocarbons were strongly bound with soil particles or had no input into the soil system. Therefore, chlorinated hydrocarbons were generally immobilized in the dredged material/soil leaching fluid systems.

Influence of various leaching solutions

124. In an attempt to evaluate the effects on leachate quality of various fluids that in the field may have an opportunity to contact dredged material, different leaching fluids were passed through laboratory lysimeter columns for a period of 3 months. For purposes of comparison, these experimental columns contained the same dredged material and interfacing soil. The leaching fluids used were as follows:

<u>Environmental Simulation</u>	<u>Leaching Fluid</u>
acid rainfall	distilled water adjusted to pH 4.5 with SO_2 gas
groundwater intrusion	50 mg/l as CaCO_3 , hardness and 50 mg/l as CaCO_3 , alkalinity added to distilled water
sanitary landfill	actual sanitary landfill leachate
marsh/bog	75 mg/l of a characterized fulvic acid solution in distilled water

Experimental results concerning the effects of various leaching fluids on the migration of constituents in dredged material/interfacing soil systems show that dredged material and soil are important reservoirs and/or sinks for various constituents, and the effects of various leaching fluids were usually negligible. It was found that the controlling mechanisms for the migration of constituents were largely regulated by the dredged material or soil and not by the various leaching fluids. As a result, the effect of various leaching fluids on the migration of constituents was limited. Only the migration trends or levels of a few parameters, e.g., total organic carbon, alkalinity,

and soluble calcium, appeared to be slightly altered by the characteristics of the leaching fluids.

Field Leachate Study

General trends

125. Through field investigations at four upland dredged material disposal sites, it was determined that the potential impact from leachates on water quality is a function of the physicochemical nature of the disposed dredged material, site-specific groundwater hydrogeological patterns, and environmental conditions of the area surrounding the site. In general, the field study found that chloride, potassium, sodium, calcium, magnesium, total organic carbon, alkalinity, and manganese in leachate may reach groundwater and pose potential problems should the waters be utilized for human consumption or agricultural purposes. The monitoring of the sites revealed that low concentrations of cadmium, copper, iron, mercury, lead, zinc, phosphate, and nickel may reach groundwaters. However, the detected levels should not pose water quality problems.

126. Bulk analyses (total acid digests) of underlying and offsite soils and dredged material showed that in many cases the dredged material and soil had similar characteristics. No definitive relationships were found, however, between total composition of the soil samples and constituent concentration levels in leachate and groundwater samples. Hydrogeological models of the four case study sites are illustrated in Figure 4; each case study site presented a somewhat unique hydrogeological system.

Characteristics of dredged material/soil

127. A comprehensive review of potential sampling devices was conducted prior to the field study. After a careful evaluation of existing literature, tubes constructed of 10-cm-diam PVC plastic pipe were selected as the most appropriate sampling device for collection of dredged material and soil samples. At each of the four case

study sites, dredged material and soil samples were obtained from locations that would provide insight for both lateral and vertical physical and chemical stratification. In general, the examination of vertical and lateral differentiations failed to reveal any systematic changes. For any particular parameter, both increases and decreases in values occurred at different sampling locations, e.g., background, on-site, or monitoring wells, as well as at different depths within each site. This is most likely due to the stratification caused by intermittent disposal activities.

128. Particle size, bulk density, and hydraulic conductivity. Comparison of the particle size analysis of predredge bottom sediments or influent samples with sediments collected at several depths within a given disposal area suggests that the material in upland disposal areas is more sandy than the original dredged bottom sediments. It was hypothesized that the reason for the differentiation was that the finer particles tend to be carried with the effluent-discharge to the receiving waters due to insufficient residence time and/or turbulence. In this study all but one dredged material sample contained more than 50 percent sand with many consisting of more than 90 percent sand. Based upon this information, it appears that fine-grained dredged material placed in upland sites cannot be guaranteed to be "self-sealing," as might occur if fine particles stayed within the site and became a barrier to leachate migration.

129. An extremely wide range of particle size distributions was observed. The site with the highest clay content averaged 20 percent clay for dredged material samples and 24 percent clay for the native soil away from the site. In many cases, dredged material and offsite soils exhibited similar particle size characteristics.

130. In most instances, dredged material samples had analyzed moisture content levels that were close to or exceeded equivalent values of moisture (i.e., the smallest value to which water content can be reduced by gravity). This suggests a potential groundwater recharge situation. The physical conditions for moderate leachate flow were present at many sampling points in all the case study sites.

131. Bulk densities of the disposed dredged material were measured. There appeared to be no definitive correlation among the vertical samples although those samples obtained from similar depths had similar bulk densities. For the types of dredged material analyzed, bulk densities varied between 1.2 to 2.2 g/cc. These values generally correlated well with the observed particle size distributions.

132. Hydraulic conductivity measurements strongly indicated heterogeneity within and among the upland dredged material disposal sites. A range of values covering five orders of magnitude was observed. Hydraulic conductivity of underlying site soils varied between 7.3×10^{-5} and 4.5×10^{-3} cm/sec; for the various types of dredged material, values varied between 1.5×10^{-8} and 3.0×10^{-3} cm/sec. The wide diversity of hydraulic conductivity within a particular site points to the potential for channeling of flow along more permeable strata.

133. Cation exchange capacity. As would be expected, this study found a wide range of cation exchange capacities for the dredged material analyzed. In general, the mean values for the cation exchange capacities reflected the relative particle textures at each site. However, linear regression analyses showed that only one site exhibited a high correlation between cation exchange capacities and clay content. This was attributed to differences in total organic carbon content and specific clay mineralogy. The cation exchange capacity levels ranged from 0.54 meq/100 g at one particular saline sediment disposal site to 84 meq/100 g at the freshwater dredged material disposal site.

134. Eh and pH. As would be expected, a wide range of Eh (oxidation-reduction potential) values and a relatively small range of pH values were reported for both offsite soils and dredged material at different depths. Levels of pH in offsite soils were found to be slightly more acidic than onsite dredged material at each of the case study sites. The mean pH value for the soils was 6.5 while that of dredged material was 7.0. This probably means that mobilization of constituents in dredged material should not be caused by fluctuations in

pH levels since they are so constant. Eh values ranged from -233 to +343 mV for dredged material and from -82 to +368 mV for offsite soils. Very little correlation was found between Eh, pH, and total organic carbon in the dredged material. The low levels of Eh (reduced conditions) found in many dredged material samples may tend to mobilize certain trace metals; iron and manganese particularly tend to increase in the solution phase under reducing conditions.

135. Total organic carbon. Average onsite total organic carbon values for the dredged material ranged from a low of 0.27 percent at one of the saline sites to a high of 3.8 percent at the freshwater dredged material disposal site. At two of the disposal sites for sediments from saline environs, offsite soils had higher average total organic carbon values than the onsite dredged material. The reverse was true at the other two case study sites. If total organic carbon is present in dredged material leachates, it may serve to mobilize certain trace metals from the underlying or adjacent soils. The presence of total organic carbon in leachates will depend to a large extent on the concentration gradient of total organic carbon in the dredged material and offsite soil. It is desirable that this concentration differential be minimal.

136. Nitrogen compounds. Total Kjeldahl (organic and ammonia) nitrogen was analyzed in both offsite soil and onsite dredged material samples. In general, Kjeldahl nitrogen values were significantly higher in the dredged material samples than in the offsite soils. The average onsite total nitrogen values ranged from a low of 269 mg/kg to a high of 3170 mg/kg. Both extreme values were in samples from two saline dredged material disposal sites. The high total Kjeldahl nitrogen levels in dredged material, in comparison with the soils, point to the potential for migration of nitrogen species, especially through conversion of organic nitrogen to ammonia nitrogen and particularly to nitrate nitrogen.

137. Phosphorus compounds. Total phosphorus was analyzed in both offsite soils and onsite dredged material. With the exception of one site, the various types of dredged material contained significantly

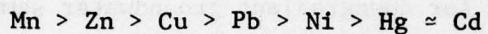
higher levels of phosphorus than offsite soils. The one exception was a saline dredged material disposal site, which had comparable levels of total phosphorus in both offsite soils and dredged material samples. The dredged material with the highest total phosphorus content came from the freshwater site; values averaged 1700 mg/kg. At the three saline dredged material disposal sites, total phosphorus values averaged 1490, 1360, and 1280 mg/kg in the dredged material. These high levels of phosphorus create concentration gradients between the onsite dredged material and offsite soils.

138. Oil and grease. Analysis of oil and grease samples showed that at three sites the dredged material samples averaged three times the values in offsite soils. This shows that a potential concentration gradient exists between the dredged material and offsite soil. The exception was the saline dredged material disposal site on the East Coast. At that site both types of samples (offsite soils and dredged material) contained comparable quantities of oil and grease. High values in the soils at that site could have been partly attributed to natural oils and organic matter derived from the overlying marsh. The lowest average dredged material oil and grease value was 610 mg/kg with the highest value being 2320 mg/kg. Both of these extreme average values were from saline dredged material disposal sites. It is not expected that oil and grease will be a problem in leachates due to the low solubility in water of most oil and grease compounds.

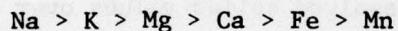
139. Chlorinated hydrocarbons. Analysis of chlorinated hydrocarbons was performed for both the offsite soils and dredged material. In general, the dredged material contained higher levels of chlorinated hydrocarbons than the offsite soil samples. Most samples contained at least some of the chlorinated hydrocarbon species studied at detectable levels. No direct correlation was found between dredged material/soil total organic carbon values and chlorinated hydrocarbon levels. Total DDT concentrations ranged from below detection limits to 850 mg/kg in both types of samples. The average values for DDT in dredged material samples ranged from 44 mg/kg at the freshwater sediment disposal

site to 96 mg/kg at one saline dredged material disposal site. These same sites also contained the lowest and highest average PCB values, namely, 0.14 and 0.58 mg/kg. Average values for dieldrin ranged from 0.8 to 2.3 mg/kg; these average values were both from saline dredged material disposal sites. There were no chlorinated hydrocarbons detected in any soluble phase samples, indicating that there is little potential for their mobilization by leachates.

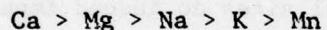
140. Metal analyses. Metals were analyzed in soluble, exchangeable, and total fractions of both the offsite soils and dredged material. In general, iron was the highest in the total sample analyses, followed by sodium, potassium, magnesium, and calcium. The remaining metals followed roughly the following order:



In the soluble and exchangeable fractions, no particular concentration trends were found for the metals other than for manganese. The major ions in the soluble phase were generally ranked as follows:



Ions in the exchangeable phase were as follows:



From these data, it appears that the major ions are most likely to be mobilized by leachates. Iron and manganese may also present problems.

Characteristics of leachates and groundwater

141. In the field investigation of the impact on groundwater of leachates from upland dredged material, various types of water samples were obtained and analyzed for selected constituents. Samples were obtained from within the site and directly below the site, as well as from background and downgradient groundwater regimes. Sampling locations were judiciously selected to enable evaluation of the vertical and horizontal migration of constituents. Water samples were obtained from unsaturated zones with the use of ceramic porous cup suction soil water samplers; samples were withdrawn from groundwaters by suction from PVC plastic wellpoints. The collected water samples were then filtered through 0.05μ membrane filters for analysis of soluble phase constituents.

142. Total dissolved solids. At three of the case study sites where total dissolved solids data were available, downgradient total dissolved solids values were higher than background samples. In all cases, onsite monitoring wells produced water samples that contained higher total dissolved solids levels than either background or offsite monitoring well samples. This is a general indication that constituents are leaching from the sites and being diluted by groundwater flow patterns. In comparing onsite average total dissolved solids values for each site, it was found that these values closely reflect the salinity of each site. Average onsite values ranged from 1870 mg/l at the freshwater dredged material disposal site to 17,000 mg/l at one of the sites containing saline sediments. The increase in average total dissolved solids values for downgradient groundwater samples over background samples ranged from 20 to 320 percent; both the low and high increases were at sites that contained saline dredged material. The freshwater disposal site exhibited a 98 percent increase in average downgradient total dissolved solids values over the average background total dissolved solids values. These values indicate that upland dredged material disposal sites do indeed impact total dissolved solids levels in underlying groundwaters.

143. Chloride. Onsite chloride concentrations ranged from an average of 167 mg/l for the freshwater sediment disposal site to 8333 mg/l at one of the saline sites. At one saline site it was found that chloride levels decreased significantly between two sampling periods. It was speculated that the reason for this decrease was dilution caused by the heavy rainfall that occurred between the sampling periods. Chloride values at the same site exhibited marked increases when additional dredged material was placed in the site prior to the scheduled sampling period.

144. The general concentration pattern for chloride showed the lowest levels in the background groundwater samples, peaking in groundwater samples collected from below the sites, then decreasing in the offsite monitoring wells as groundwater migrated laterally. The

decreases in concentration in the offsite monitoring wells were attributed to groundwater dilution of the more saline leachates. No chloride migration was observed at the freshwater site. High levels of chloride in leachates from saline sites may present water quality problems should the leachates and/or impacted groundwaters be discharged to freshwater environments, including groundwater sources used for domestic or agricultural purposes.

145. Sodium and potassium. This field investigation found highly correlatable data for sodium and potassium. Average onsite sodium concentrations were 4310, 2690, and 1485 mg/l at the three saline dredged material disposal sites and 110 mg/l at the freshwater site. At each of the case study sites, background monitoring well samples produced significantly lower sodium values than the onsite wells, which indicates a potential for leaching. This sodium leaching potential was reinforced by the fact that downgradient groundwater concentrations were an order of magnitude greater than the background concentrations. Similar trends were observed for potassium. Average onsite concentrations for potassium were 256, 81, and 168 mg/l at the three sites containing saline dredged material. The freshwater site produced an average on-site value for potassium of 29 mg/l. For both sodium and potassium it was speculated that dilution was the primary mechanism for the concentration decreases between onsite and offsite monitoring well concentrations. It was also speculated that, to a lesser extent, ion exchange regulated the sodium and potassium levels. It is felt that the high levels of these elements found in samples could present water quality problems, although most pertinent water quality standards do not contain criteria for sodium and potassium (Tables 3 and 4).

146. Calcium and magnesium. Results from the field study indicate that land disposal of both saline and freshwater dredged material can serve as a source for calcium and magnesium to groundwaters. At each of the case study sites, both calcium and magnesium levels were higher in the onsite water samples than in background samples, indicating a potential for their migration. Levels in downgradient

samples were generally lower than in onsite samples, yet higher than in background samples. This revealed that calcium and magnesium migration was occurring. Average onsite calcium values ranged from 140 to 390 mg/l at the saline dredged material disposal sites. A value of 356 mg/l was found at the freshwater site for onsite calcium. Average onsite magnesium values ranged from 174 to 728 mg/l at the three saline sediment disposal sites while the freshwater site averaged 71 mg/l. At the saline disposal sites, calcium levels in onsite water samples exhibited increases of 32 to 422 percent over background averages, while similar samples showed an average increase of 71 percent at the freshwater site. Magnesium levels also increased significantly; average increases of 142 to 178 percent over background levels were recorded at the saline sites as well as an increase of 87 percent at the freshwater site.

147. The data indicate that calcium migration was controlled by ion exchange and dissolution of calcite. The possible mechanisms for controlling magnesium migration were hypothesized to be dissolution of magnesium carbonate solids, such as nesquehonite and hydromagnesite, as well as ion exchange. While most pertinent water quality standards do not contain criteria for calcium and magnesium, it is felt that the high levels of these elements in samples could present water quality problems.

148. Alkalinity. This field study found an extremely wide range of alkalinity values in water samples; values ranged from nil to nearly 2000 mg/l. The lowest alkalinity levels occurred at one site that had background pH levels of below 5. At sites where environmental conditions were not so acidic, higher levels of alkalinity were detected in down-gradient groundwater samples than in background samples, indicating a movement of alkalinity. In these instances, water samples from onsite monitoring wells contained higher alkalinity levels than offsite monitoring well samples, indicating a dilution by underlying groundwaters. Onsite alkalinity values averaged 196 mg/l at the saline dredged material disposal sites and 590 mg/l at the freshwater site. A correlation between total organic carbon and alkalinity in many samples

indicated that biological action is probably influencing alkalinity levels. It was also speculated that dissolution of calcite regulated alkalinity levels. Most water quality standards do not contain specific criteria for alkalinity. EPA aquatic life criteria limit alkalinity changes to 75 percent of natural conditions. High levels analyzed in certain samples demonstrate that high alkalinity levels may present water quality problems following upland dredged material disposal.

149. Total organic carbon. Total organic carbon exhibited trends in mobility similar to alkalinity in two of the case study sites. However, one saline dredged material disposal site had higher average total organic carbon values in offsite monitoring well samples than in on-site samples. This site-specific occurrence was attributed to total organic carbon release from an adjacent salt marsh. The other field sites exhibited higher average total organic carbon values in down-gradient groundwater samples than in background samples, and also the highest total organic carbon levels in average onsite water samples. This indicates that total organic carbon is migrating from certain sites. Average onsite total organic carbon values ranged from 72 to 304 mg/l at the saline dredged material disposal sites and averaged 244 mg/l at the freshwater site. No water quality standards exist for total organic carbon. It is strongly felt however that total organic carbon could be a problem because of the potential for complexation of various trace metals with organic molecules. Total organic carbon in leachates may serve to mobilize some trace metals in soils. It is desirable to place dredged material in sites whose soils contain total organic carbon levels similar to the dredged material.

150. Phosphorus compounds. This study found no apparent correlation between soluble and total phosphate concentrations in the water samples analyzed. Most of the soluble phosphate levels analyzed in the water samples were very low. It was hypothesized that adsorption onto clay minerals controlled phosphate to low levels. In the samples analyzed for soluble phosphorus, values ranged from nil to 0.11 mg/l. It is not expected that phosphorus species will pose problems

to underlying groundwaters when dredged material is disposed into upland areas.

151. Cadmium. The results of water sample analyses for cadmium indicate that cadmium should not be a problem following upland disposal of dredged material. At both saline and freshwater sediment disposal sites, the average cadmium concentration at monitoring stations showed no increase over background water conditions. At one saline site, background cadmium levels were higher than the onsite values. This differentiation was attributed to site environmental conditions. Average cadmium concentrations were generally very low; the exception to this was at the site that had acidic soil conditions (background stations in a salt marsh). Average values for cadmium in onsite monitoring well water samples ranged from 1.3 to 58 $\mu\text{g/l}$ at the saline dredged material disposal sites; the highest values were at the salt marsh site displaying acidic groundwater conditions. An average value for cadmium of 0.8 $\mu\text{g/l}$ existed in onsite water samples from the freshwater site. It was speculated that cadmium migration could be controlled by low-solubility cadmium solids and adsorption onto clay minerals. The EPA public water supply criteria allow a cadmium concentration of 0.01 mg/l (Table 3). Most of the analyzed water samples had values that were considerably below that figure. The study concluded that the observed cadmium levels should not present groundwater quality problems.

152. Copper. The results of this field study indicate that copper can migrate to groundwater in leachates from upland dredged material disposal sites. Data from all sites showed higher levels of copper in downgradient monitoring station samples over background samples. Onsite samples were generally higher than downgradient samples. The exception to this was the site located near a salt marsh (slightly acid, high organic soil conditions). As would be expected, through complexation with organic matter, copper levels were high in all types of samples from that site. Onsite average copper concentrations ranged from 26 to 231 $\mu\text{g/l}$ (high values from the salt marsh site) at the saline dredged material disposal sites. The average onsite copper value at

the freshwater site was 19 $\mu\text{g/l}$. The study speculated that the mobility of copper is controlled by complexation with organic molecules. The EPA public water supply criteria level for copper is 1 mg/l (Table 3). Most of the analyzed water samples had values that were near or below that figure. This field study concluded that copper in leachates from land disposal sites should not present problems for groundwater quality.

153. Nickel. The results of this study showed that nickel concentrations were among the highest of the analyzed trace metals. Comparison of average values from the various types of water samples revealed that nickel was being leached into groundwaters underlying the disposal sites. Average onsite nickel levels ranged from 42 to 420 $\mu\text{g/l}$ at saline sediment disposal areas; the freshwater site had an average onsite nickel level of 128 $\mu\text{g/l}$. In general, nickel concentrations were reduced by about 50 to 75 percent during horizontal movement of leachate away from the sites. This study speculated that nickel migration was controlled by a number of mechanisms, the predominant one probably being transformation of nickel solids. Existing public water supply criteria do not contain standards for nickel levels. There is a value for nickel of 100 $\mu\text{g/l}$ in marine water quality criteria (Table 4). Most of the analyzed nickel values were below this level in downgradient water samples. The field study concluded that the observed nickel migration and attenuation should not present water quality problems.

154. Lead. Most of the water samples in this study contained only minute quantities of lead. Concentration levels ranged from below 1 $\mu\text{g/l}$ (found at all four sites) to a high of 80 $\mu\text{g/l}$ which was measured in two samples from the same saline dredged material disposal site. Comparison of the background and downgradient samples revealed that lead is not migrating from the sites into groundwaters to any large extent. Average soluble lead concentrations ranged from 1.9 to 14 $\mu\text{g/l}$ at the three saline dredged material disposal sites. The high average value was attributed to environmental conditions, which include a lead processing industry immediately adjacent to the disposal site.

Concentrations of lead ranged from less than 1 to 3 $\mu\text{g/l}$ at the freshwater sediment disposal site. It was speculated that adsorption by iron and manganese oxides and clay was responsible for the low levels of lead measured in this study. Virtually all analyzed lead levels were below the EPA public water supply criterion of 50 $\mu\text{g/l}$ (Table 3). The study concluded that lead in disposal area leachates should not present groundwater degradation problems.

155. Zinc. This study found no generalized trends for zinc migration. At one saline site, average zinc levels in downgradient monitoring well samples (72 $\mu\text{g/l}$) were intermediate between high levels in onsite wells (600 $\mu\text{g/l}$) and low levels in background samples (24 $\mu\text{g/l}$). At the freshwater site, average zinc values for onsite and downgradient well samples were approximately equal (450 $\mu\text{g/l}$); this figure was 40 percent less than the average background zinc level. At the site adjacent to the lead processing company and acid marsh, zinc levels were not dependent on location, and all samples showed high concentrations (average of 2440 $\mu\text{g/l}$). This study hypothesized that zinc migration was controlled by multiple factors, including adsorption, biological effects, solids transformation, complexation, and dilution; adsorption appeared to be the dominant mechanism. Virtually all analyzed zinc levels were well below the EPA drinking water standard of 5 mg/l (Table 3). It was concluded that zinc in leachates from upland disposal sites should rarely be a problem for groundwater quality.

156. Iron. This study found that iron levels were related to Eh and pH levels, i.e., lower Eh and pH levels produced higher iron values. The results of this study showed that iron is not migrating from the sites. High iron levels in downgradient and background water samples seemed to be indicative of local environmental conditions. In certain cases average iron values were higher in downgradient samples than on-site samples. This situation could be caused by the mobilization of iron from soil by organics in the leachates.

157. At one saline dredged material disposal site, average downgradient iron values (0.078 mg/l) were higher than both average background values (0.038 mg/l) and average onsite values (0.054 mg/l). At

the saline site located in an acid salt marsh, average background iron levels (538 mg/l) were much higher than both average onsite values (3.5 mg/l) and average downgradient values (3.7 mg/l). The average onsite iron value at the freshwater site was 0.055 mg/l which is 70 percent less than the average background level. The field study concluded that iron transport was influenced by Eh and pH of the surrounding site environment, solid transformation, and complexation with organic matter. The EPA drinking water standards limit for iron of 0.3 mg/l makes iron a potential water quality problem. Many onsite and downgradient water samples contained iron levels that greatly exceeded the drinking water standard for iron. It was also concluded that levels of iron found in the background well water samples could pose water quality problems. Iron migration with dredged material leachates did not appear to be a problem; localized environmental conditions, which may or may not be related to disposal activities, produced localized high levels of soluble phase iron.

158. Manganese. This field study found that manganese behaved quite similar to iron and was similarly influenced by pH and Eh. It was speculated that the high solubility of reduced manganese solids (e.g. manganese carbonate), low solubility of most manganese oxides, and the strong influence of extraneous inorganic and organic complexes on the formation of manganese solids resulted in the great variability in observed manganese concentrations. Generally there was no correlation between concentration and sample location. However, many samples had levels which were high enough to cause potential water quality problems. Average onsite manganese levels ranged from 15.5 to 218 mg/l at sites containing saline dredged material. Average downgradient and background values were either higher or lower than onsite values depending upon the site. Manganese values averaged 1.21 mg/l in the onsite samples at the freshwater site; in this instance both average downgradient and background values were higher. This study concluded that manganese migration was mainly controlled by localized pH and Eh conditions within a site. Most of the manganese concentrations in onsite and downgradient

water samples greatly exceeded the EPA public water supply criteria of 0.05 mg/l (Table 3). It was concluded that manganese will present water quality problems following upland dredged material disposal.

159. Mercury. In this study the analyzed mercury levels were quite consistent when compared to those of other trace metals. Many of the concentrations were less than 1 $\mu\text{g/l}$. Average onsite mercury values ranged from 0.34 to 0.45 $\mu\text{g/l}$ at the saline dredged material disposal sites; the freshwater site had an average onsite mercury value of 0.45 $\mu\text{g/l}$. The field study concluded that complexation and adsorption with soil organic matter and inorganic sediment fractions accounted for the low mercury levels. In addition it was concluded that mercury in leachates from upland disposal areas should not pose groundwater quality problems. Virtually all concentrations of mercury in the water samples were well below the 2 $\mu\text{g/l}$ limit set by the EPA drinking water standards (Table 3).

PART IV: CONCLUSIONS AND RECOMMENDATIONS

Conclusions

160. In order to adequately evaluate the effects of effluent discharge and leachate migration from upland dredged material disposal areas on water quality, laboratory and field results have been compared to existing water quality standards. These comparisons are presented in Tables 2-4. The water quality impact from upland disposal of dredged material has been assessed based on the beneficial use of the receiving waters. The major areas of beneficial use include public water supplies, crop irrigation, and preservation of aquatic life in both freshwater and marine water environments. Effluent discharge from actively used disposal sites as well as potential long-term leachate generation has been considered.

161. The data presented in Table 2 as well as detailed discussions in Part II of this report on the effluents from active disposal sites clearly indicate the dilemma of meeting appropriate water quality criteria. In most cases, soluble concentrations of most chemical constituents are quite low. However, total concentrations are generally high due to the presence of suspended and colloidal particles.

162. The data given in Tables 3 and 4 show similarities in the average soluble constituent values for both the laboratory and field studies. Of particular note is the excellent correlation between field and laboratory results for the major ions (both saline and freshwater dredged material). Concerning the trace metals, it appears that the average laboratory lysimeter data for a given metal were often five to ten times higher than comparable data from the field study. This variance may have been partly due to the use of different interfacing soils in the laboratory simulation than the soils underlying actual case study sites. Since many of the similarities between the laboratory and the field leachate data were for troublesome contaminants, a laboratory column leachate study should give a rough estimation of the potential impact on groundwater of placing a particular dredged sediment in an

upland disposal area. It appears that the column studies present a worst case situation for many trace metals.

163. Based on the obtained data, the following conclusions were drawn:

a. Impact of effluent discharge on receiving waters:

- (1) The concentrations of ammonia nitrogen, total phosphorus, chlorinated hydrocarbons, and most of the trace metals in unfiltered effluent water samples fail to meet most water quality criteria for beneficial use of receiving waters.
- (2) Most soluble phase contaminant levels, with the notable exceptions of manganese and ammonia nitrogen, are below most present criteria. Thus, efficient removal of suspended solids is necessary to meet these guidelines.
- (3) Actively growing, thick vegetation in disposal areas seems to improve the removal efficiencies of most contaminants by improving the removal of suspended solids, as well as greatly improving the removal of soluble nutrients (e.g., ammonia and orthophosphate). Therefore, the disposal of dredged material in vegetated areas may be a plausible method for improved effluent water quality; the cycling of low-solids effluents through marsh or agricultural lands may result in additional removal of soluble nutrients while stimulating productivity. The placement of brackish water dredged material or effluents on salt-sensitive vegetation is not desirable.
- (4) Salinity levels resulting from the disposal of brackish water dredged material should seriously affect the beneficial uses of surface receiving waters used for public water supplies and irrigation.

b. Impact of leachates on groundwaters:

- (1) Disposal of brackish water dredged material in upland disposal areas may render subsurface water unsuitable for public water supply or irrigation use.
- (2) Most of the leachates from the freshwater disposal site contained ammonia and nitrate nitrogen, iron, and manganese in concentrations that exceed drinking water standards.

- (3) Based on data from three sites, leachates migrating from brackish water dredged material may contain ammonia, iron, and manganese exceeding marine water criteria. The site-specific nature of soil attenuation and groundwater dilution are the major factors in determining the possible impact.

Recommendations

164. Based on the preceding conclusions, it is recommended that existing upland dredged material disposal sites be monitored to ascertain impact on underlying groundwaters. This monitoring may take the form of analysis of major ions as well as other minor constituents in water samples withdrawn from downgradient wells; major ions may be used to indicate movement of leachate fronts from upland disposal sites through groundwater regimes.

165. It is recommended that criteria be developed for upland disposal based on the chemical characteristics of dredged material, proposed site groundwater hydrology, and prevailing environmental conditions for both saline and freshwater upland dredged material disposal sites. To aid in the selection of potential sites, it is recommended that evaluatory procedures be developed that may apply to any given site.

166. It is desirable that guidelines for evaluation of potential upland disposal sites be developed in a stepwise progression that will not require complete execution of the total program in assessing the suitability of any particular site. A site-specific evaluation may include the following items (listed in the most logical order for most areas):

- a. Socioeconomic acceptability in the area of potential problems, including aesthetic factors.
- b. Beneficial uses planned for the receiving waters.
- c. Hydrogeological survey, including depth to local and regional groundwater tables, groundwater flow patterns, salinity profiles, geological formations, and soil type(s).

- d. Potential of the soil for retarding leachate movement (e.g., soil permeability determinations and homogeneity of soil profile).
- e. Economic and engineering feasibility for use of artificial barriers or treatments to control or correct potential effluent or leachate problems.
- f. Similarity of site soil with the predredging bottom sediment, both chemically and physically; compatibility of soil interstitial water with predredging bottom sediment interstitial water.
- g. Development of laboratory column or other predictive testing and/or conduct field monitoring to determine the compatibility of effluents with surface receiving waters, and the compatibility of leachates with subsurface receiving waters (groundwaters).

167. The major purpose of a stepwise progression is that site evaluatory procedures should be developed that will be germane to any given site, i.e., if it is found that a site is underlain by an extensive layer of uninterrupted impervious bedrock between the site and the groundwater, then further site evaluation may not be necessary. It is felt that a detailed, extensive evaluation of every potential site may cause unwarranted expenditures. In certain instances, an upland site may be deemed acceptable if interstitial waters from proposed sediments to be dredged and site soils are similar. If this were the case, further evaluation of the site in question may be unnecessary. Development of a soil attenuation/mobilization model that may be used to predict the concentration of various constituents in leachates, based on dredged material and underlying soil characteristics, is recommended only if other evaluatory factors indicate a potential problem. Under "real world" conditions, a predictive model (if proven reliable) may be desirable over laboratory simulations because of the time frame involved in performing laboratory leachate studies.

168. In conclusion, upland dredged material disposal must be a carefully controlled practice. In some instances, it may be an acceptable disposal method; under other conditions, upland dredged material disposal may not be acceptable at all, or artificial effluent or leachate control systems may have to be installed prior to site utilization.

Existing sites should be periodically monitored to obtain a better understanding of the behavior of leachate and contaminant migration.

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Table 1
Descriptions of the Dredging and Confined Land Disposal Areas and Operations Studied in Task 2D

General Location	Work Unit	Location of Dredging Site	Location of Disposal Site	Predominant Dredged Sediment	Description of Disposal Area			Groundwater Flow Direction	Treatment Modes
					Effluent/Runoff Discharge Site	Size of Diked Area hectares	Shape		
Sayreville, N. J.	2001 2002 2005	South channel of Raritan River (brackish water)	National Lead Industries Disposal Area No. 4, adjacent to river	Dark grey silt	Raritan River (brackish surface water)	1.8	~16	Roughly circular; divided into three equal compartments	Overland flow Ponding Vegetation interaction Internal dikes
Houston, Tex.	2001 2002 2005	Houston Ship Channel (brackish water)	East half of Clinton disposal area, about 1.5 km inland from channel	Fine reddish sand and black silt, heavily impregnated with weathered crude oil	Hunting Bayou-Houston Ship Channel, 3 km east of dredging site (fresh to brackish surface water)	11.3	~91	Rectangular	Artificial system: Radially from southeast corner of site
Grand Haven, Mich.	2001 2002 2005	Grand River (fresh water)	Verplank's Coal & Dock Co. disposal areas, Ferrysburg, Mich., adjacent to river	Fine sand with some oily and high organic fines	Grand River (fresh surface water)	2.4	~2.4	Roughly rectangular	Low gradient groundwaters flowing southeast through the site and into the river
Wilmington, N. C.	2001	Anchorage Basin, Cape Fear River (fresh to brackish water)	Eagle Island disposal area, between Cape Fear and Brunswick Waterways	Dark grey silt and clay	Brunswick Waterway (fresh to brackish surface water)	21.2	~162	Circular	Overland flow Minor ponding Vegetation interaction
Richmond, Va.	2001	James River, Deepwater Terminal (fresh water)	Disposal area on east bank of James River	Coarse sand and gravel, some light brown silts; dark grey silt on last day	James River (fresh surface water)	28	~14	Long and narrow; divided into three equal square compartments	Overland flow Shallow ponding Cross-dikes
Lake Charles, La.	2001	Calcasieu River near Lake Calcasieu (brackish water)	Disposal area No. 22, 15 river km south of Lake Charles, on dredged material island between the ship channel and lake	Dark grey to reddish-brown mixed silt, clay, and fine sand; some oily sediments	Calcasieu River (fresh to brackish surface water)	75	~61	Roughly rectangular	Overland flow Ponding

(Continued)

Note: 2001, influent-effluent characterization (field study); 2002, leachate characterization (laboratory lysimeter study); 2004, influent-effluent fractionation (field study);
2005, leachate characterization (field study).

Table 1 (Concluded)

General Location	Work Unit	Location of Dredging Site	Location of Disposal Site	Predominant Dredged Sediment	Effluent/Runoff Discharge Site	Description of Disposal Area			Treatment Modes
						Size of Diked Treatment Area (hectares)	Size of Groundwater Flow Area (hectares)	Direction	
Seattle, Wash.	2D01	Duwanish Waterway	Old wastewater treatment plant sludge lagoon area north of dredging site, 60 m from waterway	Black silt-clay	Duwanish Waterway (brackish surface water)	0.75	0.75	Rectangular; divided into two equal rectangular compartments, each 46 by 85 by 4.5 m	Ponding Cross dike
Vicksburg, Miss.	2D01	Brown Lake, WES (fresh water)	Adjacent to upper end of lake	Light grey silt with light brown crust	Durden Creek-Brown Lake (fresh surface water)	2.0	2.0	Rectangular; divided into two equal compartments	Ponding Natural cross-dike
Southport, N. C.	2D01	Elizabeth River (brackish to marine)	East end of Oak Island, with dikes adjacent to Intra-coastal Waterway	Black silt-clay	Atlantic Intra-coastal Waterway (brackish surface water)	19.5	~18	Elongated	Overland flow Ponding Vegetation interaction
Mobile, Ala.	2D02	Mobile River (brackish water)	North Blakeley Island, adjacent to Mobile River	Black silt-clay	Mobile River (brackish surface water)	--	--	Roughly square	
Mobile, Ala.	2D04 2D05	Mobile River (brackish water)	Pinto Island (between Mobile River and Mobile Bay)	Mixed dark-grey to brown silt and sand	Confluence of Mobile River and Mobile Bay (brackish to fresh surface water)	26	~19	Elongated	Radial flow from center of site
Detroit, Mich.	2D04	Main channel of Rouge River (fresh water)	Grassy Island	Dark brown silty-clay; some oil	Detroit River near entrance to Lake Erie (fresh surface water)	12	~11	Rectangular (north half used)	Overland flow Ponding Vegetation interaction

Table 2
Comparisons of Brackish and Freshwater Effluent Constituent Concentrations* with
Marine and Freshwater Criteria

Parameter	Marine and Freshwater Criteria										Effluent Average									
	Agriculture (Irrigation)					Freshwater (Aquatic Life)					Marine Water (Aquatic Life)					Brackish Water Sites		Freshwater Sites		
	EPA	NAS	NTAC	EPA	NAS	EPA	NAS	NTAC	EPA	NAS	NTAC	EPA	NAS	NTAC	Total	Soluble	Total	Soluble		
pH	4.5-	4.5-	4.5-	6.0-	6.5-	6-9	5-9	6-8.5	6.5-	6.5-	8.5	8.5	8.5	8.5	7.5	—	7.4	—		
Dissolved Oxygen	—	—	—	—	4.0	5.0	—	—	—	6.0	—	5.0	4.8	—	—	7.7	—	7.4	—	
Salinity	—	—	500-5000	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	650	
Nitrate-N	—	—	—	—	0.02	0.02	—	0.5	0.5	0.5	0.4	0.4	0.4	—	17.3	15.1	13.5	13.5	8.64	
Ammonia-N	—	—	—	—	—	—	—	0.1	0.1	0.1	0.01	0.01	0.01	—	21.7	0.44	3.48	3.48	0.08	
Total-P	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0.42	—	0.505	—	—	
DDT	—	—	—	—	0.000002	0.000002	—	—	—	—	—	—	—	—	—	—	—	—	—	
PCB's	—	—	—	—	0.000002	0.000001	—	—	0.000001	—	—	—	—	—	0.15	—	0.93	—	—	
Arsenic	0.1-2	0.1-2	1-20	—	—	—	—	0.05	—	—	—	0.2	0.2	—	0.079	0.046	0.32	0.32	0.0003	
Cadmium	0.01-	0.01-	0.005-	0.03	0.03	—	0.01	0.1	0.1	0.1	0.1	0.01	—	—	0.032	0.046	0.088	0.088	0.0008	
Chromium	0.1-1	0.1-1	5-20	0.05	0.05	0.02	0.05	0.5	0.5	0.1	0.05	—	—	—	0.032	0.025	0.26	0.005	—	
Copper	0.2-5	0.2-5	0.2-5	—	—	—	—	1	1	1	—	—	—	—	0.48	0.021	0.57	0.0049	—	
Iron	5-20	5-20	—	—	—	—	—	0.3	0.3	0.3	0.3	0.3	0.3	0.3	423	1.21	54.3	54.3	0.10	
Lead	5-10	5-10	5-20	0.03	0.03	—	0.05	0.05	0.05	0.05	0.05	0.01	—	1.82	0.0026	0.291	0.0008	—	—	
Manganese	0.2-10	0.2-10	2-20	—	—	—	0.05	0.05	0.05	0.1	0.02	—	—	13.2	1.95	1.30	1.30	0.29	—	
Mercury	—	—	—	0.05	0.05	—	0.002	0.002	—	0.1	0.1	—	—	0.0083	0.0012	0.019	0.019	0.0002	—	
Nickel	0.2-2	0.2-2	0.5-2	—	—	—	—	—	—	—	0.1	—	—	—	0.416	0.015	0.272	0.272	0.0037	
Selenium	0.02	0.02	0.05	—	—	—	—	0.01	—	—	0.01	0.005	—	—	1.89	0.0026	0.16	0.0005	—	
Vanadium	—	—	10	—	—	—	—	—	—	—	0.5	0.1	—	—	1.15	0.013	0.22	0.22	0.0039	
Zinc	—	—	2-10	5-10	—	—	—	5	5	5	0.1	—	—	—	2.63	0.061	1.28	1.28	0.008	

* All concentrations are in mg/l (parts per million), except pH.

Table 3
**Comparisons of Freshwater Dredged Material Leachate Constituent Concentrations* with
 Public Water Supply Criteria**

Parameter	Public Water Supply Criteria				Leachate			
	EPA (Regulation)		NAS (Proposed)	NTAC (Proposed)	Laboratory Values		Field Values	
	Average	Range	Average	Range	Average	Range	Average	Range
TOC	--	--	--	--	251	21-650	211	50-370
Alkalinity (as CaCO ₃)	--	--	--	--	754	42-1639	734	264-1120
Ammonia-N	0.5	0.5	0.5	1.9	0.30-11.6	--	--	--
TKN**	--	--	--	27	15.6-40	--	--	--
Nitrate-N	10	10	10	12.2	<0.01-30.4	--	--	--
Orthophosphate-P	--	--	--	0.03	<0.01-0.107	--	--	--
pH	5-9	5-9	6-8.5	7.1	5.95-8.45	6.9	6.3-7.2	31-680
Sodium	--	--	--	53.9	7-208	114	7-110	7-110
Potassium	--	--	--	74.5	22-320	34	50-480	50-480
Calcium	--	--	--	136	28-320	261	20-93	20-93
Magnesium	--	--	--	53.9	2.9-215	66	64-1050	64-1050
Chloride	250	250	--	197	53-860	185	N11-0.830	N11-0.830
Iron	0.3	0.3	0.3	5.8	0.2-23	0.125	0.002-2.7	0.0007-0.001
Manganese	0.05	0.05	0.05	13.3	0.1-30	0.76	0.001-0.059	0.001-0.001
Cadmium	0.01	0.01	0.01	0.002	N11-0.0021	0.00079	0.0001-0.001	0.0001-0.001
Copper	1	1	1	0.0045	N11-0.01	0.015	0.0004	0.0004
Mercury	0.002	0.002	--	0.00083	N11-0.0025	0.0008	0.0001-0.030	0.0001-0.030
Lead	0.05	0.05	0.05	0.0057	N11-0.030	0.019	0.0001-0.080	0.0001-0.080
Zinc	5	5	5	0.071	0.01-0.18	0.019	0.0001-0.080	0.0001-0.080

* All concentrations are in mg/l (parts per million), except pH.

** Total Kjeldahl nitrogen.

Table 4
Comparisons of Dredged Material Leachate Constituent Concentrations* with Marine Receiving Water Criteria

Parameter	Marine Receiving Water (Aquatic Life) Criteria						Leachate Field Values Range	
	EPA	NAS	NTAC	State of California				
				50% of Time	100% of Time	Average		
TOC	--	--	--	--	--	4.48	136-770 79 N11-240	
Alkalinity (as CaCO_3)	--	--	--	--	--	534	42-1296 484 45-977	
Ammonia-N	0.4	0.4	0.05 × 96 hr LC_{50}	0.4	0.6	3.55	0.4-7.65 -- --	
TKN**	--	--	--	--	--	47.9	18.1-98.9 -- --	
Orthophosphate-P	0.01	0.005	--	--	--	0.015	0.01-0.043 -- --	
pH	6.5-8.5	6.5-8.5	6.5-8.5	0.2 unit change	0.2 unit change	7.1	5.84-8.12 7.0 5.6-8.3	
Sodium	--	--	--	--	--	1431	22-6220 2151 20-5800	
Potassium	--	--	--	--	--	181	25-970 108 17-200	
Calcium	--	--	--	--	--	774	32-2270 173 29-350	
Magnesium	--	--	--	--	--	227	7-739 237 12-550	
Chloride	--	--	--	--	--	4710	410-10,500 3190 16-9220	
Iron	0.3	0.1	--	--	--	12.5	1.0-144 0.30 0.003-0.210	
Manganese	0.1	0.1	--	0.1	0.1	48.1	0.1-122 17 0.4-53.6	
Cadmium	0.1	0.01	--	0.02	0.03	0.0105	0.001-0.086 0.004 0.0001-0.046	
Copper	0.01 × 96 hr LC_{50}	0.01 × 96 hr LC_{50}	--	0.2	0.3	0.0125	0.001-0.140 0.041 0.001-0.225	
Mercury	0.1	0.1	--	0.0001	0.0002	0.0076	N11-0-.0025 0.0003 N11-0.001	
Lead	0.02 × 96 hr LC_{50}	0.02 × 96 hr LC_{50}	--	0.1	0.2	0.0948	0.002-0.485 0.0014 N11-0.0037	
Zinc	0.1	--	--	0.3	0.5	0.128	0.040-0.340 0.410 0.006-3.51	

* All concentrations are in mg/l (parts per million), except pH.

** Total Kjeldahl nitrogen.

Table 5
Total Solids Content and Particle Size Distribution

		Nonfilterable (Suspended) Solids Percent Weight		Particle Size, Clay		Percent Silt		Percent Sand		Coulter Counter Particle Size (Median Size, μ)								
Brackish Water Sites**																		
Freshwater Sites																		
Influent	Average	6.35	12.3	53.8	34.1	12.1	—	—	—	0.98								
	Range	5.65-6.7	6.26-18.5	42-65	26-50	6-25	—	—	—	0.91-1.05								
Effluent	Average	1.6	0.577	59.5	32.5	8	—	—	—	2.55								
	Range	0.009-2.5	0.015-1.22	58-61	30-35	4-12	—	—	—	2.0-3.1								
Background water	Average	0.11	0.011	—	—	—	—	—	—	5.4								
	Range	—	0.0036-0.027	—	—	—	—	—	—	—								
Brackish Water Sites**																		
Influent	Average	15.2	12.5	16.8	48.3	35	—	—	—	1.02								
	Range	3.49-23.5	1.66-23.3	11-23	24-73	5-65	—	—	—	0.89-1.15								
Effluent	Average	0.14	0.147	—	—	—	—	—	—	6.3								
	Range	0.06-0.297	0.046-0.248	—	—	—	—	—	—	3.75-8.9								
Background water	Average	0.027	0.0015	—	—	—	—	—	—	6.55								
	Range	0.011-0.043	0.001-0.002	—	—	—	—	—	—	5.5-7.6								

* Values exclude total dissolved solids.

** Averages and ranges are calculated from average site values.

Table 6
Physicochemical Parameters

		<u>Salinity, ‰</u>	<u>Dissolved Oxygen, mg/l</u>	<u>Temperature, °C</u>	<u>Alkalinity, mg/l</u>	<u>pH</u>	<u>CEC, mg/l</u>
Brackish Water Sites*							
Influent	Average	14.6	2.7	19.2	461	7.3	58
	Range	2.8-25.5	0.65-4.9	8.5-27.9	151-1024	6.6-7.9	28-81
Effluent	Average	16.2	4.8	19.8	344	7.5	72
	Range	2.6-21.3	2.2-7.9	9.9-28.4	178-631	6.8-7.9	12-121
Background water	Average	7.6	8.0	17.8	67	6.7	--
	Range	0.5-21.0	7.6-8.3	7.8-27.7	17-112	5.5-7.6	--
Freshwater Sites							
Influent	Average	0.37	6.1	13.7	278	6.9	35
	Range	0.15-0.7	3.0-7.8	1.5-24.3	82-505	6.7-7.1	10-69
Effluent	Average	0.65	7.7	15.2	190	7.4	66
	Range	0.1-1.25	2.4-12.5	0.8-26.1	61-244	7.2-7.7	--
Background water	Average	N11	9.3	16.3	157	7.1	--
	Range	--	7.0-11.5	3.5-29.0	41-290	6.7-7.6	--

* Averages and ranges are calculated from average site values.

Table 7
Total Sulfides and Nitrogen Compounds

		Ammonia Nitrogen ($<0.45 \mu$), mg/l	Organic Nitrogen ($<0.45 \mu$), mg/l	Nitrate Nitrogen ($<0.45 \mu$), mg/l	Total Sulfides mg/l
<u>Brackish Water Sites*</u>					
Influent	Average	21.6	5.57	0.084	54.6
	Range	4.83-50.6	1.5-14.8	0.01-0.27	3.8-122
Effluent	Average	15.1	3.03	0.16	4.8
	Range	1.25-49.7	0.78-7.1	0.01-0.36	<0.02-18.4
Background water	Average	0.25	0.73	0.27	0.75
	Range	0.01-0.62	0.19-1.35	0.01-0.68	<0.02-2
<u>Freshwater Sites</u>					
Influent	Average	18.7	3.03	0.36	17.2
	Range	6.95-38.5	1.6-5.82	0.2-0.59	3.0-38.4
Effluent	Average	8.64	2.01	0.52	1.0
	Range	4.08-13.0	1.20-2.7	0.11-1.33	0.2-1.7
Background water	Average	0.26	0.45	1.10	1.7
	Range	0.05-0.54	0.25-0.65	0.01-1.98	1.4-1.9

* Averages and ranges are calculated from average site values.

Table 8
Phosphorus Compounds

		Total Phosphorus (Slurry), mg/l	Total Phosphorus (<0.45 μ), mg/l	Orthophosphate Phosphorus (<0.45 μ), mg/l
<u>Brackish Water Sites*</u>				
Influent	Average	160	1.28	1.08
	Range	74.3-309	0.14-6.12	0.10-4.31
Effluent	Average	21.7	0.55	0.31
	Range	0.25-48.3	0.1-1.30	0.09-1.00
Background water	Average	0.29	0.25	0.05
	Range	0.07-0.86	0.06-0.50	0.03-0.09
<u>Freshwater Sites</u>				
Influent	Average	138	0.08	0.075
	Range	24.9-347	0.06-0.11	0.06-0.09
Effluent	Average	3.48	0.11	0.09
	Range	0.15-8.66	0.02-0.20	0.09
Background water	Average	0.41	0.11	0.13
	Range	0.15-0.76	0.02-0.15	0.11-0.15

* Averages and ranges are calculated from average site values.

Table 9
Organic Carbon and Oil and Grease

	Total Oil and Grease mg/l	Oil and Grease (<0.45 μ), mg/l	Total Organic Carbon, mg/l	Organic Carbon (<0.45 μ), mg/l
Brackish Water Sites*				
Influent	Average 552	9.7	3130	28.6
	Range 456-618	2.4-21.3	19.4-6060	8-107
Effluent	Average 37	5.3	152	20.4
	Range 5.9-73.5	--	5.0-475	6.4-68
Background water	Average 13.5	1.1	9.3	15.0
	Range 0.13-47.2	--	4-14	3.2-43
Freshwater Sites				
Influent	Average 2650	3.8	788	16.4
	Range 39.5-5260	2.5-5	63-1820	10-28
Effluent	Average 13.5	2.0	16.7	13.4
	Range 3.9-25	--	10-24	6-28
Background water	Average 19.2	1.1	9.5	16.3
	Range 6.4-32	--	9-10	6-31

* Averages and ranges are calculated from average site values.

Table 10
Chlorinated Pesticides and PCB's

		DDE		DDD		DDT		Total*		Total PCB's mg/l	
		<u>Brackish Water Sites**</u>				<u>Total*</u>		DDT			
		Average	DDE	DDD	DDT	Total	DDT	mg/l			
Influent	Average	0.47	0.54	2.35	3.36	3.75	3.75				
	Range	0.035-0.80	<0.01-1.41	<0.01-8.7	--	--				1.28-5.88	
Effluent	Average	0.36	0.06	<0.01	0.42	0.15	0.15				
	Range	<0.01-0.96	<0.01-0.25	<0.01	--					0.0006-0.72	
Background water	Average	0.62	<0.01	<0.01	0.62	<0.1	<0.1				
	Range	<0.01-1.85	<0.01	<0.01	--					0.00001-0.3	
<u>Freshwater Sites</u>											
Influent	Average	14.9	11.5	3.95	30.35	26.0	26.0				
	Range	0.37-57.1	<0.01-45.2	<0.01-14.3	--	--				4.75-80.1	
Effluent	Average	0.275	0.06	0.17	0.505	0.93	0.93				
	Range	0.02-0.50	<0.01-0.25	<0.01-0.60	--					<0.1-2.55	
Background water	Average	0.15	0.05	0.02	0.22	<0.1	<0.1				
	Range	<0.01-0.32	<0.01-0.14	<0.01-0.06	--					<0.1-0.21	

* Combined total concentration of DDE, DDD, and DDT.

** Averages and ranges are calculated from average site values.

Table 11
Trace Metal Concentrations and Removal Efficiencies

	Fe	Mn	Zn	Cd	Cu	Ni	Pb	Hg	Total Trace Metals, mg/l	Cr	Ti	V	As	Se
Brackish Water Sites*														
Influent	3640	59.4	23.0	0.78	4.99	4.4	14.2	0.035	--	4.29	3.98	0.73	3.10	
Effluent	423	13.2	2.63	0.032	0.48	0.416	1.82	0.0083	0.032	1.55	1.15	0.079	1.89	
Background water	12.0	0.10	0.561	0.0054	0.11	0.308	0.083	0.0031	0.020	0.011	0.32	0.008	--	
Percent** removal	88	78	89	96	90	91	87	76	--	64	71	89	39	
Effluent/Background water	35	132	4.7	6.0	4.4	1.4	22	3	1.6	141	3.6	10	--	
Freshwater Sites														
Influent	4080	82.5	24.4	1.63	28.1	9.83	10.2	0.073	63.8	8.3	3.97	4.55	4.95	
Effluent	54.3	1.30	1.28	0.088	0.57	0.272	0.291	0.019	0.26	0.26	0.22	0.32	0.16	
Background water	2.75	0.23	0.365	0.0016	0.13	0.016	0.018	0.00043	0.013	--	0.029	0.004	0.008	
Percent removal	99	98	95	95	98	97	97	74	-100	97	95	93	97	
Effluent/Background water	20	5.6	3.5	5.5	4.4	17	16	44	20	--	7.6	80	20	

* Averages are calculated from average site values.
 ** Disposal area removal efficiency:

$$\frac{\text{Influent-Effluent}}{\text{Influent}} \times 100$$

(Continued)

Table 11 (Concluded)

	Fe	Mn	Zn	Cd	Soluble Trace Metals (<0.45 μ), mg/l				Tl	V	As	Se				
	<u>Brackish Water Sites*</u>															
<u>Influent</u>																
Influent	3.72	2.86	0.065	0.0047	0.020	0.016	0.0032	0.0017	--	0.018	0.032	0.0033				
Effluent	1.21	1.95	0.061	0.0046	0.021	0.015	0.0026	0.0012	0.025	0.016	0.013	0.0046				
Background water	0.46	0.079	0.040	0.0042	0.011	0.015	0.0021	0.0010	--	0.0001	0.0040	0.0006				
Percent** Removal	67	32	6	2	+5 [†]	6	19	29	--	6	28	83				
Effluent	2.6	25	1.5	1.1	1.9	1.0	1.2	1.2	--	160	3	5.8				
Background water												4.6				
<u>Freshwater Sites</u>																
Influent	3.00	1.00	0.022	0.0018	0.0053	0.0069	0.0017	0.0002	0.004	1.83	0.0041	0.0004				
Effluent	0.10	0.29	0.008	0.0007	0.0049	0.0037	0.0008	0.0002	0.005	1.45	0.0039	0.0003				
Background water	0.10	0.030	0.007	0.001	0.0046	0.0043	0.0007	0.0002	0.003	--	0.004	0.0003				
Percent Removal	97	71	64	58	8	46	53	0	+25 [†]	21	5	25				
Effluent	1.0	10	1.1	0.75	1.1	0.85	1.1	1.0	1.7	--	1.0	1.0				
Background water												--				

* Averages are calculated from average site values.

** Disposal area removal efficiency:

$$\frac{\text{Influent-Effluent}}{\text{Influent}} \times 100$$
.

† Percent increase.

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Table 12
Ratios of Effluent Trace Metal Concentrations to
EPA/NAS/NTAC Criteria*

	Agriculture* (Irrigation)		Freshwater (Aquatic Life)	Freshwater (Public Water)	Marine Water (Aquatic Life)
	Brackish	Fresh			
Total Trace Metals					
As	0.8	3	--	6	0.4
Cd	6	18	3	9	3
Cr	0.3	3	13	5	0.6
Cu	2	3	--	0.6	--
Fe	85	11	--	180	8500
Pb	0.4	0.06	10	6	180
Mn	66	7	--	26	660
Hg	--	--	0.1	10	0.08
Ni	2	1.4	--	--	4
Se	95	8	--	16	380
V	0.12	0.02	--	--	12
Zn	1.3	0.6	--	0.3	26
Soluble Trace Metals (<0.45 μ)					
As	0.05	0.003	--	0.01	0.02
Cd	0.9	0.2	0.03	0.1	0.5
Cr	0.3	0.05	0.3	0.1	0.5
Cu	0.1	0.02	--	0.005	--
Fe	0.2	0.02	--	0.3	24
Pb	0.0005	0.0002	0.03	0.02	0.3
Mn	10	1.5	--	6	98
Hg	--	--	0.001	0.1	0.01
Ni	0.08	0.02	--	--	0.15
Se	0.13	0.03	--	0.05	0.5
V	0.001	0.0004	--	--	0.1
Zn	0.03	0.004	--	0.002	0.6

* Calculated from minimum criteria.

In accordance with letter from DAEN-RDC, DAEN-ASI dated 22 July 1977, Subject: Facsimile Catalog Cards for Laboratory Technical Publications, a facsimile catalog card in Library of Congress MARC format is reproduced below.

Chen, Kenneth Y

Confined disposal area effluent and leachate control (laboratory and field investigations / by Kenneth Y. Chen ... et al., University of Southern California, Los Angeles, Calif., and Environmental Laboratory, U. S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.) Vicksburg, Miss. : U. S. Waterways Experiment Station ; Springfield, Va. : available from National Technical Information Service, 1978.

94 p. : ill. ; 27 cm. (Technical report - U. S. Army Engineer Waterways Experiment Station ; DS-78-7)

Prepared for Office, Chief of Engineers, U. S. Army, Washington, D. C.

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TA7.W34 no.DS-78-7